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Air Quality Criteria for Lead (First External Review Draft)

Volume I of II

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Air Quality Criteria for Lead

Volume I

National Center for Environmental Assessment-RTP Office
Office of Research and Development
U.S. Environmental Protection Agency
Research Triangle Park, NC

DISCLAIMER

This document is a first external review draft being released for review purposes only and does not constitute U.S. Environmental Protection Agency policy. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

PREFACE

National Ambient Air Quality Standards (NAAQS) are promulgated by the United States Environmental Protection Agency (EPA) to meet requirements set forth in Sections 108 and 109 of the U.S. Clean Air Act. Sections 108 and 109 of the Clean Air Act require the EPA Administrator (1) to list widespread air pollutants that reasonably may be expected to endanger public health or welfare; (2) to issue air quality criteria for them that assess the latest available scientific information on nature and effects of ambient exposure to them; (3) to set "primary" NAAQS to protect human health with adequate margin of safety and to set "secondary" NAAQS to protect against welfare effects (e.g., effects on vegetation, ecosystems, visibility, climate, manmade materials, etc); and (5) to periodically review and revise, as appropriate, the criteria and NAAQS for a given listed pollutant or class of pollutants.

Lead was first listed in the mid-1970's as a "criteria air pollutant" requiring NAAQS regulation. The scientific information pertinent to Lead NAAQS development that was available at the time was assessed in the EPA document *Air Quality Criteria for Lead*; published in 1977. Based on the scientific assessments contained in that 1977 lead air quality criteria document (1977 Lead AQCD), EPA established a 1.5 $\mu g/m^3$ (90-day average) Lead NAAQS in 1978.

To meet Clean Air Act requirements noted above for periodic review of criteria and NAAQS, newly available scientific information published since the 1977 Lead AQCD was assessed and discussed in a revised Lead AQCD and Addendum published in 1986 and in a Supplement to the 1986 AQCD/Addendum published by EPA in 1990. A 1990 Lead Staff Paper, prepared by EPA's Office of Air Quality Planning and Standards (OPQPS), drew upon key findings and conclusions from the 1986 Lead AQCD/Addendum and 1990 Supplement (as well as other OAQPS-sponsored lead exposure/risk analyses) in posing options for the EPA

Administrator to consider with regard to possible revision of the Lead NAAQS. However, EPA decided not to revise the lead NAAQS at that time.

The purpose of this revised Lead AQCD is to critically evaluate and assess the latest scientific information that has become available since the literature assessed in the above 1986 Lead AQCD/Addendum and 1990 Supplement, with the main focus being on pertinent new information useful in evaluating health and environmental effects of ambient air lead exposures. This includes discussion in this document of information regarding: the nature, sources, distribution, measurement, and concentrations of lead in the environment; multimedia lead exposure (via air, food, water, etc.) and biokinetic modeling of contributions of such exposures to concentrations of lead in brain, kidney, and other tissues (e.g., blood and bone concentrations, as key indices of lead exposure).; characterization of lead health effects and associated exposure-response relationships; and delineation of environmental (ecological) effects of lead. This First External Review Draft of the revised Lead AQCD mainly assesses pertinent literature published or accepted for publication through June, 2004.

The present First External Review Draft (dated December 2005) of the revised Lead AQCD is being released for public comment and review by the Clean Air Scientific Advisory Committee (CASAC) to obtain comments on the organization and structure of the document, the issues addressed, the approaches employed in assessing and interpreting the newly available information on lead exposures and effects, and the key findings and conclusions arrived at as a consequence of this assessment. Public comments and CASAC recommendations will be taken into account in making appropriate further revisions to this document for incorporation into a Second External Review Draft of the document to be released in early 2006 for further public comment and CASAC review. Public comments and CASAC advice received on the Second External Review Draft materials will then be taken into account in incorporating further revisions into the final version of this Lead AQCD, which is to be completed and issued by October 1, 2006. Evaluations contained in the present document will be drawn on to provide inputs to associated Lead Staff Paper prepared by EPA's Office of Air Quality Planning and Standards (OAQPS), which will pose options for consideration by the EPA Administrator with regard to proposal and, ultimately, promulgation of decisions on potential retention or revision, as appropriate, of the current Lead NAAQS.

Preparation of this document was coordinated by staff of EPA's National Center for Environmental Assessment in Research Triangle Park (NCEA-RTP). NCEA-RTP scientific staff, together with experts from and academia, contributed to writing of document chapters. Earlier drafts of document materials were reviewed by scientists from other EPA/units and by non-EPA experts in several public peer consultation workshops held by EPA in July/August 2005.

NCEA acknowledges the valuable contributions provided by authors, contributors, and reviewers and the diligence of its staff and contractors in the preparation of this draft document.

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(First External Review Draft)

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Abbreviations and Acronyms

αFGF	α-fibroblast growth factor
AA	arachidonic acid; atomic absorption
AALM	All Ages Lead Model
AAS	atomic absorption spectroscopy
ACBP	Achenbach Child Behavior Profile
ACE	angiotensin converting enzyme
ACR	acute-chronic ratio
ACSL	Advanced Continuous Simulation Language
ADC	analog-digital converter
ADHD	Attention Deficit/Hyperactivity Disorder
ADP	adenosine dinucleotide phosphate
AEA	*N*-arachidonylethanolamine
AF	absorption fraction
2-AG	2-arachidonylglycerol
A horizon	uppermost layer of soil (litter and humus)
AHR	aryl hydrocarbon receptor
ALA	δ-aminolevulinic acid
ALAD	δ-aminolevulinic acid dehydratase
ALAS	aminolevulinic acid synthase
ALS	amyotrophic lateral sclerosis
ALT	alanine aminotransferase; alanine transferase
AMD	activity mean diameter
AMP	adenosine monophosphate
ANF	atrial natriuretic factor
ANOVA	analysis of variance
AP-1	activated protein-1
APE	apurinic endonuclease
AQCD	Air Quality Criteria Document
ASV	anode stripping voltammetry
ATP	adenosine triphosphate
ATP142	sodium-potassium adenosine triphosphate α2
ATPase	adenosine triphosphate synthase

ATSDR	Agency for Toxic Substances and Disease Research
AVS	acid volatile sulfide
AWQC	ambient water quality criteria
β	beta-coefficient; slope of an equation
βFGF	β-fibroblast growth factor
6-β-OH-cortisol	6-β-hydroxycortisol
B cell	B lymphocyte
BAEP	brainstem auditory-evoked potentials
BAER	brainstem auditory-evoked responses
BAF	bioaccumulation factor
BCF	bioconcentration factor
BLL	blood lead level
BLM	biotic ligand model
BMDM	bone marrow-derived macrophages
BMI	body mass index
BMP-6	bone morphogenic protein-6
BRHS	British Regional Heart Study
BTQ	Boston Teacher Questionnaire
BUN	blood urea nitrogen
BW, bw	body weight
CA	chromosomal aberration
^{45}Ca	calcium-45 radionuclide
CA1	cornu ammonis 1 region of hippocampus
CA3	cornu ammonis 3 region of hippocampus
CAA	Clean Air Act
Ca-ATPase	calcium-dependent adenosine triphosphatase
$^{43}CaCl_2$	calcium-43 radionuclide-labeled calcium chloride
$CaCO_3$	calcium carbonate
CaEDTA	calcium disodium ethylenediaminetetraacetic acid
CAL	calcitonin
cAMP	cyclic adenosinemonophosphate
$CaNa_2$ EDTA	calcium disodium ethylenediaminetetraacetic acid
CANTAB	Cambridge Neuropsychological Testing Automated Battery
CASAC	Clean Air Scientific Advisory Committee

CBCL	Achenbach Child Behavior Checklist
CCE	Coordination Center for Effects
CDC	Centers for Disease Control and Prevention
CEC	cation exchange capicity
CESD, CES-D	Center for Epidemiologic Studies Depression (scale)
cGMP	cyclic guanosine-3',5'-monophosphate; cyclic guanylylmonophosphate
CI	confidence interval
CLRTAP	Convention on Long-range Transboundary of Air Pollution
CMC	criterion maximum concentration
CMI	cell-mediated immunity
CNS	central nervous system
ConA	concanavalin A
COX-2	cyclooxygenase-2
CP	coproporphyrin
CPT	current perception threshold
CRAC	calcium release activated calcium reflux
CRI	chronic renal insufficiency
CSF	cerebrospinal fluid
CSF-1	colony-stimulating factor-1
CTH	cystathionine gamma-lyase
CTL	cytotoxic T lymphocyte
CuZn-SOD	copper and zinc-dependent superoxide dismutase
CYP2A6	cytochrome P-2A6
CYP3A4	cytochrome P-3A4
CYP450	cytocrome P-450
DET	diffusive equilibrium thin films
DFS	decayed or filled surfaces, permanent teeth
dfs	covariate-adjusted number of caries
DGT	diffusive gradient thin films
DL	detection limit
DMEM	Dulbecco’s modified eagle medium
DMFS	decayed, missing, or filled surfaces, permanent teeth
DMSA	2,3-dimercaptosuccinic acid

DMTU	dimethyl thio urea
DNA	deoxyribonucleic acid
DNTC	diffuse neurofibrillary tangles with calcification
DOC	dissolved organic carbon
DOM	dissolved organic matter
DOS	Disc Operating System
DPASV	differential pulse anode stripping voltammetry
DTH	delayed type hypersensitivity
DTPA	diethylenetriaminepentaacetic acid
dw	dry weight
E_2	estradiol
E_b	electron binding energies
EBE	early biological effect
EC_{50}	effect concentration for 50% of test population
eCB	endocannabinoid (e.g., 2-arachidonylglycerol [2-AG] and *N*-arachidonylethanolamine [AEA])
ECF	extracellular fluid
Eco-SSL	ecological soil screening level
EDRF	endothelium-derived relaxing factor
EDS	energy dispersive spectrometers
EDTA	ethylenediaminetetraacetic acid
EEG	electroencephalogram
EGF	epidermal growth factor
eNOS	endothelial nitric oxide synthase
EOD	explosive ordnance disposal
EP	erythrocyte protoporphyrin
EPA	U.S. Environmental Protection Agency
EPMA	electron probe microanalysis
EPT	macroinvertebrates from the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) group
EqP	equilibrium partitioning (theory)
ERG	electroretinogram
EROD	ethoxyresorufin-O-deethylase
ESP	electrostatic precipitator

ESRD	end-stage renal disease
ET	endothelein; essential tremor
EXAFS	Extended X-ray absorption fine structure
EXANES	Extended X-ray absorption near edge spectroscopy
FAV	final acute value
FDA	Food and Drug Administration
FEF	forced expiratory flow
FEV_1	forced expiratory volume in one second
FIAM	free ion activity model
FMLP	N-formyl-L-methionyl-L-leucyl-L-phenylalanine
f_{OC}	fraction organic carbon
FPLC	fast protein liquid chromatography
FR	federal register
FSH	follicle stimulating hormone
FT3	free triiodothyronine
FT4	free thyroxine
FVC	forced vital capacity
G6PD	glucose-6-phosphate dehydrogenase
GABA	gamma aminobutyric acid
GAG	glycosaminoglycan
GCI	General Cognitive Index
GD	gestational day
GDP	guanosine diphosphate
GEE	generalized estimating equations
GFAAS	graphite furnace atomic absorption spectroscopy
GFR	glomerular filtration rate
GH	growth hormone
GI	gastrointestinal
GM	geometric mean
GMAV	genus mean acute value
GMP	guanosine monophosphate
GnRH	gonadotropin releasing hormone
g_{OC}	grams organic carbon)
GP	gross productivity

GPEI	glutathione S-transferase P enhancer element
GRP78	glucose-regulated protein 78
GSD	geometric standard deviation
GSD_i	individual geometric standard deviation
GSH	glutathione; reduced glutathione
GSIM	gill surface interaction model
GSSG	oxidized glutathione
GST	glutathione transferase; glutathione S-transferase
GTP	guanosine triphosphate
H^+	acidity
H_2O_2	hydrogen peroxide
Hb	hemoglobin
HBEF	Hubbard Brook Experimental Forest
Hct	hematocrit
HDL	high-density lipoprotein (cholesterol)
HFE	hemochromatosis gene
HFF	human foreskin fibroblasts
HH	hydroxylamine hydrochloride
HHC	hereditary hemochromatosis
5-HIAA	5-hydroxyindoleacetic
HOME	Home Observation for Measurement of Environment
HPLC	high-pressure liquid chromatography
HQ	hazard quotient
HSI	habitat suitability index
HSPG	heparan sulfate proteoglycan
HTN	hypertension
HVA	homovanillic acid
IARC	International Agency for Research on Cancer
ICP	inductively coupled plasma
ICP-AES	inductively coupled plasma atomic emission spectroscopy
ICP-MS	inductively coupled plasma mass spectrometry
ICRP	International Commission on Radiological Protection
IDMS	isotope dilution mass spectrometry
IEC	intestinal epithelial cells

IEUBK	Integrated Exposure Uptake Biokinetic (model)
IFN	interferon (e.g., IFN-γ)
Ig	immunoglobulin (e.g., IgA, IgE, IgG, IgM)
IGF_1	insulin-like growth factor 1
IL	interleukin (e.g., IL-1, IL-1β, IL-4, IL-6, IL-12)
IMPROVE	Interagency Monitoring of Protected Visual Environments (network)
iNOS	inducible nitric oxide synthase
i.p., IP	intraperitoneal
IQ	intelligence quotient
IQR	interquartile range
IT	intrathecal
i.v., IV	intravenous
KTEA	Kaufman Test of Educational Achievement
K-XRF	K-shell X-ray fluorescence
LA	lipoic acid
LC_{50}	lethal concentration (at which 50% of exposed animals die)
LDL	low-density lipoprotein (cholesterol)
L-dopa	3,4-dihydroxyphenylalanine (precursor of dopamine)
LH	luteinizing hormone
LMW	low molecular weight
L-NAME	L-N^G-nitroarginine methyl ester
LOAEL	lowest-observed adverse effect level
LOEC	lowest-observed-effect concentration
LPO	lipid peroxide; lipid peroxidation
LPS	lipopolysaccharide
LT_{50}	time to reach 50% mortality
LTD	long-term depression
LTP	long-term potentiation
LVH	left ventricular hypertrophy
μPIXE	microfocused particle induced X-ray emission
μSXRF	microfocused synchrotron-based X-ray fluorescence
MAO	monoaminoxidase
MATC	maximum acceptable threshold concentration

MCH	mean corpuscular hemoglobin
MCHC	mean corpuscular hemoglobin concentration
MCV	mean corpuscular volume
MDA	malondialdehyde
MDA-TBA	malondialdehyde-thiobarbituric acid
MDI	Mental Development Index
MDRD	Modification of Diet in Renal Disease (study)
meso-DMSA	meso-2,3-dimercaptosuccinic acid
Mg-ATPase	magnesium-dependent adenosine triphosphatase
MHC	major histocompatibility complex
miDMSA	mono-3-methylbutane-1-yl (monoisomyl) ester of meso-2,3-dimercaptosuccinic acid
MINTEQ	thermodynamic equilibrium model
MINTEQA2	equilibrium speciation computer model
MK-801	NMDA receptor antagonist
MLR	mixed lymphocyte response
MMSE	Mini-Mental State Examination
MN	micronuclei formation
Mn-SOD	manganese-dependent superoxide dismutase
MRI	magnetic resonance imaging
mRNA	messenger ribonucleic acid
MRS	magnetic resonance spectroscopy
MSV	Moloney sarcoma virus
MT	metallothionein
MVV	maximum voluntary ventilation
N, n	number of observations
N/A	not available
NAA	N-acetylaspartate; neutron activation analysis
NAAQS	National Ambient Air Quality Standards
NAC	N-acetyl cysteine
NAD	nicotinamide adenine nucleotide
NADH	reduced nicotinamide adenine dinucleotide; nicotinamide adenine dinucleotide dehydrogenase
NADP	nicotinamide adenine dinucleotide phosphate

NAD(P)H	reduced nicotinamide adenine dinucleotide phosphate
NADS	nicotinamide adenine dinucleotide synthase
NAG	N-acetyl-β-D-glucosaminidase
Na-K-ATPase	sodium-potassium-dependent adenosine triphosphatase
NAS	Veterans' Administration Normative Aging Study
NASCAR	National Association for Stock Car Automobile Racing
NAWQA	National Water-Quality Assessment
NCBP	National Contaminant Biomonitoring Program
NCEA-RTP	National Center for Experimental Assessment Division in Research Triangle Park, NC
ND	not detected; non-detectable
NE	norepinephrine
NEPSY	Developmental Neuropsychological Assessment
NF-κB	nuclear transcription factor-κB
NHANES	National Health and Nutrition Examination Survey
NIOSH	National Institute for Occupational Safety and Health
NIST	National Institute for Standards and Technology
NK	natural killer
NMDA	N-methyl-D-aspartate
nNOS	neuronal nitric oxide synthase
NO	nitric oxide
NO_3	nitrate
NOAEL	no-observed-adverse-effect level
NOD	autoimmune diabetes prone strain of mice
NOEC	no-observed-effect concentration
NOM	natural organic matter
NOS	nitric oxide synthase
NP	net productivity
NR	not reported
NRC	National Research Council
NTP	National Toxicology Program
O_2^-	superoxide ion
OAQPS	Office of Air Quality Planning and Standards
OAR	Office of Air and Radiation

O/E	observed-expected ratio
OH	hydroxyl
1,25-OH-D	1,25-dihydroxyvitamin D
25-OH-D	25-hydroxyvitamin D
$1,25\text{-}(OH_2)D_3$	vitamin D
25-OH D_3	25-hydroxycholecalciferol
O horizon	forest floor
$ONOO^-$	peroxynitrate ion
OR	odds ratio
ORD	Office of Research and Development
p	probability value
P_{10}	probability for the occurrence of a blood lead concentration exceeding 10 µg/dL
PAD	peripheral arterial disease
PAH	polycyclic aromatic hydrocarbon
PAI-1	plasminogen activator inhibitor-1
Pb	lead
^{203}Pb	lead-203 radionuclide
^{204}Pb, ^{206}Pb, ^{207}Pb, ^{208}Pb	stable isotopes of lead-204, -206, -207, -208 respectively
PbB	blood lead; blood lead concentration
$PbCO_3$	lead carbonate
PBG-S	porphobilinogen synthase
$Pb(OH)_2$	lead hydroxide
$Pb(NO_3)_2$	lead nitrate
PbS	galena
PC12	pheochromocytoma cell
PFCs	plaque forming cells
PG	prostaglandin (e.g., PGE_2 ,PGF_2)
PHA	phytohemagglutinin A
P_i	inorganic phosphorus
PIR	poverty-income ratio
PIXE	particle induced X-ray emission
PKC	protein kinase C
PKC-α	protein kinase C α

plasma-ECF	plasma and extracellular fluid combined
PM	particulate matter
PM_{10}	combination of coarse and fine particulate matter
$PM_{2.5}$	fine particulate matter
PMN	polymorphonuclear leukocyte
PMNL	polymorphonuclear leukocyte
P5N	pyrimidine 5'-nucleotidase
PND	postnatal day
p.o., PO	per os (oral administration)
POMS	Profile of Mood States
ppb	parts per billion
ppm	parts per million
PRL	prolactin
PTH	parathyroid hormone
PTHrP	parathyroid hormone-related protein
PVC	polyvinyl chloride
PWM	pokeweed mitogen
Q	flux of air
QA/QC	quality assurance/quality control
Qco_2	flux of carbon dioxide
r	Pearson correlation coefficient
r^2	correlation coefficient
RAAS	renin-angiotensin-aldosterone system
rac-DMSA	racemic-2,3-meso-2,3-dimercaptosuccinic acid
RBA	relative bioavailablity
RBC	red blood cell; erythrocyte
RBP	retinol binding protein
RCPM	Ravens Colored Progressive Matrices
RDA	recommended daily allowance
RDW	red cell distribution
ROS	reactive oxygen species
ROS 17.2.8	rat osteosarcoma cell line
RR	relative risk
RSD	relative standard deviation

∑SEM	sum of the molar concentrations of simultaneously extracted metal
∑TU	summed of toxic units for all metals in a mixture
SAB	Science Advisory Board
SAM	S-adenosyl methionine
s.c., SC	subcutaneous
SCE	sister chromatid exchange
SD	standard deviation; Spraque-Dawley (rat)
SDS-PAGE	sodium dodecyl sulfate-polyacrylamide gel electrophoresis
SE	standard error; *Staphylococcus aureus* enterotoxin
SEM	simultaneously extracted metal; standard error of the mean
SES	socioeconomic status
SHBG	sex hormone binding globulin
SIMS	secondary ion mass spectrometry
SIR	standardized incidence ratio
SLP	synthetic leaching procedure
SMAV	species mean acute value
SMR	standardized mortality ratio
SO_2	sulfur dioxide
SOD	superoxide dismutase
SOILCHEM	chemical species equilibrium model
SRBC	sheep red blood cell
SRC	Syracuse Research Corporation
SRD	Self Report of Delinquent Behavior
SRE	sterol regulatory element
SRM	standard reference materials
SRT	simple reaction time
STORET	database for the STOrage and RETrieval of chemical, physical, and biological data
$T_{1/2}$; $t_{1/2}$	half-time
T3	triiodothyronine
T4	thyroxine
TBA	thiobarbituric acid
TBARS	thiobarbituric acid-reactive species
T_C	cytotoxic T lymphocyte

T cell	T lymphocyte
TCLP	toxic characteristic leaching procedure
TEL	tetraethyllead; tri ethyl lead
TES	testosterone
TGF	transforming growth factor (e.g., TGF-α ,TGF-β, TGF-β1)
T_H	T-helper lymphocyte
Th0	precursor T lymphocyte
Th1	T-derived lymphocyte helper 1
Th2	T-derived lymphocyte helper 2
^{232}Th	stable isotope of thorium-232
tHct	total hematocrit
tHcy	plasma total homocysteine
^{203}Tl, ^{205}Tl	stable isotopes of thallium-203 and -205, respectively
TLC	treatment of lead-exposed children
T_M	T-memory lymphocyte
TML	tetramethyllead
TNF	tumor necrosis factor (e.g., TNF-α, TNF-β1)
tPA	plasminogen activator
TPBS	Total Problem Behavior Score
TRV	toxicity reference value
TSH	thyroid stimulating hormone
TSP	total suspended particulates
TT3	total triiodothyronine
TT4	serum total thyroxine
TTR	transthyretin
TU	toxic unit
TWA	time-weighted average
TX	tromboxane (e.g., TXB_2)
^{235}U, ^{238}U	uranium-234 and -238 radionuclides
UDP	uridine diphosphate
UNECE	United Nations Economic Commission for Europe
USGS	United States Geological Survey
UV	ultraviolet
V	volume of culture

Vd	deposition velocity
VDR	vitamin D receptor
VEP	visual-evoked potential
vitamin D	1,25-dihydroxyvitamin D_3
VLDL	very low density lipoprotein (cholesterol)
VMI	visual motor integration
VSMC	vascular smooth muscle cells
w/v	weight per volume
WDS	wavelength dispersive spectrometers
WHO	World Health Organization
WIC	Women, Infants, and Children (program)
WISC-III	Wechsler Intelligence Scale for Children-III
WRAT-R	Wide Range Achievement Test-Revised
ww	wet weight
XANES	extended X-ray absorption near edge spectroscopy
XAS	X-ray absorption spectroscopy
XPS	X-ray photoelectron spectroscopy
X-rays	synchrotron radiation
XRD	X-ray diffraction
XRF	X-ray fluorescence

1. INTRODUCTION

The present document critically assesses the latest scientific information concerning health and welfare effects associated with the presence of various concentrations of lead (Pb) in ambient air, as pertinent to providing updated scientific bases for EPA's current periodic review of the National Ambient Air Quality Standards for Lead (Pb NAAQS). As such, this document builds upon previous assessments published by the U.S. Environmental Protection Agency (EPA), including: (a) the document, *Air Quality Criteria for Lead* (U.S. Environmental Protection Agency, 1977); (b) an updated revision of that Lead Air Quality Criteria Document (Lead AQCD) and an accompanying Addendum published in 1986 (U.S. Environmental Protection Agency, 1986a,b); as well as (c) an associated 1990 Supplement (U.S. Environmental Protection Agency, 1990). This document focuses on evaluation and integration of information relevant to Pb NAAQS criteria development that has become available mainly since that covered by the 1986 and 1990 criteria assessments.

This introductory chapter (Chapter 1) of the revised Lead AQCD presents: (a) background information on pertinent Clean Air Act legislative requirements, the criteria and NAAQS review process, and the history of previous Pb criteria reviews; (b) an overview of the current Pb criteria review process, associated key milestones, and projected schedule; and (c) an orientation to the general organizational structure and content of this revised Lead AQCD.

1.1 LEGAL AND HISTORICAL BACKGROUND

1.1.1 Legislative Requirements

Two sections of the Clean Air Act (CAA) govern the establishment, review, and revision of NAAQS. Section 108 (42 U.S.C. 7408) directs the Administrator of the U.S. Environmental Protection Agency (EPA) to identify ambient air pollutants that may be reasonably anticipated to endanger public health or welfare and to issue air quality criteria for them (U.S. Code, 2003a). These air quality criteria are to reflect the latest scientific information useful in indicating the kind and extent of all identifiable effects on public health or welfare that may be expected from the presence of a given pollutant in ambient air.

Section 109(a) of the CAA (42 U.S.C. 7409) directs the Administrator of EPA to propose and promulgate primary and secondary NAAQS for pollutants identified under Section 108 (U.S. Code, 2003b). Section 109(b)(1) defines a primary standard as one that, in the judgment of the Administrator, is requisite to protect the public health (see inset below) based on the criteria and allowing for an adequate margin of safety. The secondary standard, as defined in Section 109(b)(2), must specify a level of air quality that, in the judgment of the Administrator, is requisite to protect the public welfare (see inset below) from any known or anticipated adverse effects associated with the presence of the pollutant in ambient air, based on the criteria.

EXAMPLES OF PUBLIC HEALTH EFFECTS

- Effects on the health of the general population, or identifiable groups within the population, who are exposed to pollutants in ambient air
- Effects on mortality
- Effects on morbidity
- Effects on other health conditions including indicators of:
  - pre-morbid processes,
  - risk factors, and
  - disease

EXAMPLES OF PUBLIC WELFARE EFFECTS

- Effects on personal comfort and well-being
- Effects on economic values
- Deterioration of property
- Hazards to transportation
- Effects on the environment, including:
  - animals
  - climate
  - crops
  - materials
  - soils
  - vegetation
  - visibility
  - water
  - weather
  - wildlife

Section 109(d) of the CAA (42 U.S.C. 7409) requires periodic review and, if appropriate, revision of existing criteria and standards (U.S. Code, 2003b). If, in the Administrator's judgment, the Agency's review and revision of criteria make appropriate the proposal of new or revised standards, such standards are to be revised and promulgated in accordance with Section 109(b). Alternatively, the Administrator may find that revision of the standards is inappropriate and conclude the review by leaving the existing standards unchanged. Section 109(d)(2) of the 1977 CAA Amendments also requires that an independent scientific review committee be established to advise the EPA Administrator on NAAQS matters, including the scientific soundness of criteria (scientific bases) supporting NAAQS decisions. This role is fulfilled by the Clean Air Scientific Advisory Committee (CASAC), which is administratively supported by EPA's Science Advisory Board (SAB).

1.1.2 Criteria and NAAQS Review Process

Periodic reviews by EPA of criteria and NAAQS for a given criteria air pollutant progress through a number of steps, beginning with preparation of an air quality criteria document (AQCD) by the National Center for Environmental Assessment Division in Research Triangle Park, NC (NCEA-RTP), a unit within EPA's Office of Research and Development (ORD). The AQCD provides a critical assessment of the latest available scientific information upon which the NAAQS are to be based. Drawing upon the AQCD, the Office of Air Quality Planning and Standards (OAQPS), a unit within EPA's Office of Air and Radiation (OAR), prepares a Staff Paper that (a) evaluates policy implications of the key studies and scientific information contained in the AQCD; (b) presents relevant exposure and risk analyses; and (c) presents EPA staff conclusions and recommendations for standard-setting options for the EPA Administrator to consider. The Staff Paper is intended to help "bridge the gap" between the scientific assessment contained in the AQCD and the judgments required of the Administrator in determining whether it is appropriate to retain or to revise the NAAQS.

Iterative drafts of both the AQCD and the Staff Paper (as well as other analyses, such as associated exposure and/or risk assessments supporting the Staff Paper) are made available for public comment and CASAC review. Final versions of the AQCD and Staff Paper incorporate changes in response to CASAC review and public comment. Based on the information in these documents, the EPA Administrator proposes decisions on whether to retain or revise the subject NAAQS, taking into account public comments and CASAC advice and recommendations. The Administrator's proposed decisions are published in the *Federal Register*, with a preamble that delineates the rationale for the decisions and solicits public comment. After considering comments received on the proposed decisions, the Administrator makes a final decision, which is promulgated via a *Federal Register* notice that addresses significant comments received on the proposal.

Promulgated NAAQS decisions involve consideration of the four basic elements of a standard: *indicator*, *averaging time*, *form*, and *level*. The indicator defines the pollutant to be measured in the ambient air for the purpose of determining compliance with the standard. The averaging time defines the time period over which air quality measurements are to be obtained and averaged, considering evidence of effects associated with various time periods of exposure. The form of a standard defines the air quality statistic that is to be compared to the level of the

standard (i.e., an ambient concentration of the indicator pollutant) in determining whether an area attains the standard. The form of the standard specifies the air quality measurements that are to be used for compliance purposes (e.g., the 98th percentile of an annual distribution of daily concentrations; the annual arithmetic average), the monitors from which the measurements are to be obtained (e.g., one or more population-oriented monitors in an area), and whether the statistic is to be averaged across multiple years. These basic elements of a standard are the primary focus of the staff conclusions and recommendations posed in the Staff Paper and are explicitly specified in the ensuing NAAQS rulemaking, building upon the policy-relevant scientific information assessed in the AQCD and on the policy analyses contained in the Staff Paper. These four elements taken together determine the degree of public health and welfare protection afforded by the NAAQS.

1.1.3 Regulatory Chronology

In 1971, U.S. EPA promulgated national ambient air standards for several major "criteria" pollutants (see Federal Register, 1971), but did not include lead among them at that time. Later, on October 5, 1978, the EPA promulgated primary and secondary NAAQS for lead, under Section 109 of the CAA (43 FR 46258), as announced in the Federal Register (1979). The primary standard and the secondary standard are the same: 1.5 $\mu g/m^3$ as a quarterly average (maximum arithmetic mean averaged over 90 days). The standards were based on the EPA's 1977 Air Quality Criteria for Lead (U.S. Environmental Protection Agency, 1977).

In 1986, the EPA published a revised Air Quality Criteria Document for Lead (U.S. Environmental Protection Agency, 1986a). The 1986 AQCD assessed newly available scientific information on the health and welfare effects associated with exposure to various concentrations of lead in ambient air, based on literature published through 1985. The 1986 document was principally concerned with the health and welfare effects of lead, but other scientific data were also discussed in order to provide a better understanding of the pollutant in the environment. Thus, the 1986 document included chapters that discussed the atmospheric chemistry and physics of the pollutant; analytical approaches; environmental concentrations; human exposure and dosimetry; physiological, toxicological, clinical, and epidemiological aspects of lead health effects; and lead effects on ecosystems. An Addendum to the 1986 Lead AQCD was also published along with it (U.S. Environmental Protection Agency, 1986b). Subsequently,

a Supplement to the 1986 Lead AQCD/Addendum was published by EPA in 1990 (U.S. Environmental Protection Agency, 1990a). That 1990 Supplement evaluated still newer information emerging in the published literature concerning (a) lead effects on blood pressure and other cardiovascular endpoints and (b) the effects of lead exposure during pregnancy or during the early postnatal period on birth outcomes and/or on the neonatal physical and neuropsychological development of affected infants and children.

The evaluations contained in the 1986 Lead AQCD/Addendum and the 1990 Supplement provided scientific inputs to support decision-making regarding periodic review and, as appropriate, revision of the Lead NAAQS; and they were drawn upon by EPA's Office of Air Quality Planning and Standards in preparation of an associated OAQPS Lead Staff Paper (U.S. Environmental Protection Agency, 1990b). However, after consideration of evaluations contained in these documents, EPA chose not to propose revision of the Lead NAAQS.

Changes in relative contributions of various lead sources and exposure pathways to human exposures in the United States, and EPA actions to reduce such exposures, provide important background for this current lead criteria and NAAQS review. Since 1978, the amount of lead emitted into the air nationally has markedly declined. For example, as illustrated in Chapters 2 and 3 of this document, from 1982 to 2002 lead emissions into the air decreased by 93% and the average air quality concentration of lead decreased by 94% from 1983 to 2002 (http://www.epa.gov/airtrends/lead2.html). Total lead emissions into the air decreased from about 220,000 tons in 1970 to less than 4,000 in 1999. This decline is mainly attributable to EPA's regulatory efforts to reduce the content of lead in gasoline (see, for example, 50 FR 9386), which substantially altered basic patterns of air lead emissions in the United States (http://www.epa.gov/airtrends/lead2.html). Emissions from stationary sources have also been greatly reduced (http://www.epa.gov/airtrends/lead2.html, Figure 2-11); but, given the even greater reductions in emissions from transportation sources, industrial processes (including smelters and battery manufacturers) now constitute a larger percentage of remaining lead emissions to the atmosphere (http://www.epa.gov/airtrends/lead2.html, Figure 2-12). In short, lead emissions into the atmosphere decreased greatly in the 1980's and 1990's, a trend that has continued on through to the present. As a consequence, airborne lead now represents only a relatively small component of total exposure to lead in the United States, such that the principal sources and pathways for U.S. lead exposure among the classically-defined most sensitive

population group (young children) involve non-inhalation pathways, e.g., ingestion of lead from deteriorating paint, food, drinking water, dust, and historically contaminated soil. While these downward trends in air lead exposures nationwide are encouraging, several important sources of air lead exposure may still persist in some localities. Lead emissions from specific stationary sources and/or reentrainment of lead-contaminated soils (including from past deposition of airborne lead) may still have significant impacts on a local level. Recognition of the multimedia nature of lead exposure of the general population has been important historically and sorting out relative contributions to total lead exposure burdens represents an important input to the current periodic Lead NAAQS review effort.

Since the 1980's, EPA has played a major, effective role in working to reduce the main sources of lead exposure for most children, including deteriorating lead-based paint, lead-contaminated dust, and lead-contaminated residential soil (http://www.epa.gov/lead/). For example, EPA has established standards for lead-based paint hazards and lead dust cleanup levels in most pre-1978 housing and child-occupied facilities, and is now developing standards for those conducting renovation activities that create lead-based paint hazards and for the management and disposal of lead-based debris (http://www.epa.gov/lead/regulation.htm). Also, EPA has developed standards for management of lead in solid and hazardous waste, continues to oversee the cleanup of lead contamination at Superfund facilities, and has issued regulations to reduce lead in drinking water (http://www.epa.gov/lead/sources.htm). Beyond taking specific regulatory actions, the Agency's Lead Awareness Program also continues to work to protect human health and the environment against the dangers of lead by conducting research and designing educational outreach efforts and materials (http://www.epa.gov/lead/).

Since the 1980's, EPA has also promulgated regulations under section 112 of the Clean Air Act (42 U.S.C. § 7412), to address emissions of lead components and other toxic pollutants from both primary lead smelters and secondary lead smelters (40 CFR Subparts X and TTT). Under section 112(d), these emission standards are to require "the maximum degree of reduction in emissions" that are "achievable." Thus, EPA promulgated section 112(d) standards for secondary lead smelters on June 23, 1995 (60 Fed. Reg. 3587) and revised them on June 13, 1997 (62 Fed. Reg. 32209), followed by promulgation of section 112(d) standards for primary lead smelters on June 4, 1999 (64 Fed. Reg. 30194).

1.2 CURRENT LEAD CRITERIA AND NAAQS REVIEW

1.2.1 Procedures and Key Milestones for Document Preparation

It is important to emphasize at the outset that development of the present document has and will continue to include substantial external (non-EPA) expert inputs and opportunities for public input through (a) public workshops involving the general scientific community, (b) iterative reviews of successive drafts of this document by CASAC, and (c) comments from the public on successive drafts. Extensive external inputs received through such reviews will help to ensure that the review of the Lead NAAQS will be based on critical assessment in this document of the latest available pertinent science.

The procedures for developing this revised Lead AQCD build on experience derived from the other recent criteria document preparation efforts, and include close coordination between NCEA-RTP and OAQPS staff, as well as with others, throughout the document preparation/review process. Briefly, the respective responsibilities for production of the document and meeting key milestones are as follows. An NCEA-RTP Lead Team has been designated as being responsible for the creation and implementation of a project plan for developing the Lead AQCD, taking into account input from individuals in other ORD units, OAQPS, and other EPA program/policy offices identified as part of the EPA Lead Work Group. The Lead Team defines critical issues and topics to be addressed by the authors and provides direction in order to focus on evaluation of those studies most clearly identified as likely being important for U.S. air standard setting purposes. Criteria document materials are authored in part by NCEA-RTP Lead Team staff with appropriate expertise in particular areas and by non-EPA consultants to EPA who are recognized experts in pertinent specific areas (e.g., lead biokinetic modeling, toxicology, epidemiology, etc.).

Key milestones for development of this Lead AQCD are listed in Table 1-1. As a first step, EPA announced on November 9, 2004 official initiation of the current periodic review of air quality criteria for lead. More specifically, under processes established in Sections 108 and 109 of the Clean Air Act, U.S. EPA began by announcing in the Federal Register (69 FR 64,926) the formal commencement of the current review process with a call for information (see Federal Register, 2004). In addition, EPA prepared a January 2005 draft Lead AQCD Work Plan, which was made available for public comment and was the subject of teleconsultation with CASAC on March 28, 2005 as a means by which to communicate the process and timeline for development

Table 1-1. Key Milestones and Projected Schedule for Development of Revised Lead Air Quality Criteria Document (Lead AQCD)[1]

Major Milestones	Target Dates
1. Literature Search	Ongoing
2. Federal Register Call for Information	November 9. 2004
3. Prepare Draft Lead AQCD Project Work Plan	Nov-Dec 2004
4. Release Draft Project Plan for Public Comment/CASAC Review	January 2005
5. Public Comment Period	Jan/Feb 2005
6. CASAC Teleconsultation on Project Work Plan	March 28, 2005
7. Workshop Drafts of Lead AQCD Chapters	May/June 2005
8. Peer Consultative-Review Workshop(s)	July/August 2005
9. Release First External Review Draft	December 1, 2005
10. Public Comment Period	Dec 2005-Feb 2006
11. CASAC/SAB Public Review Meeting *(First Ext. Rev. Draft)*	Feb. 28-Mar 1, 2006
12. Release Second External Review Draft	June 2006
13. Public Comment Period	June-July 2006
14. CASAC/SAB Public Review Meeting *(Second Ext. Rev. Draft)*	August 2006
15. Final Lead AQCD	October 1, 2006

[1] Schedule may be modified from time to time, as necessary, to reflect actual project requirements and progress, but EPA is under court order to produce a final Lead AQCD by October 1, 2006. Missouri Coalition for the Environment v. EPA, Civil Action No. 4:04-CV-00660 (ERW) (E.D. Mo. Sept. 14, 2005). Also, note that materials contributed by non-EPA authors, at times, have been and will continue to be modified by EPA staff in response to internal and/or external review comments and that EPA is responsible for the ultimate content of this Lead AQCD.

of a revised Lead AQCD. Next, expert consultants to NCEA-RTP and NCEA-RTP staff (a) carefully evaluated pertinent new studies obtained via the call for information and via ongoing literature searches conducted by NCEA-RTP information retrieval specialists and (b) prepared preliminary draft chapter materials for inclusion in this revised Lead AQCD. Those preliminary draft materials then underwent expert peer discussion at public workshops organized and conducted by NCEA-RTP in July/August, 2005. After consideration of comments received at the workshops, appropriate revisions were made in the draft materials and incorporated into this First External Review Draft of the Lead AQCD, which is now being made available for public comment (the comment period ends February 15, 2006) and CASAC review at a public

meeting scheduled for February 28-March 1, 2006 (to be announced in the Federal Register). EPA expects that, after consideration of CASAC and public comments, it will prepare a Second External Review Draft of this revised Lead AQCD for further review by the public and CASAC before completing the final version of it for issuance by October 1, 2006. Publication of the final document and its availability to the public will be announced in the Federal Register.

Drawing upon evaluations in the Lead AQCD and other lead exposure/risk analyses, the EPA's Office of Air Quality Planning and Standards (OAQPS) staff will prepare a draft Lead Staff Paper that will assess policy implications of key information in the Lead AQCD, report pertinent exposure and risk analyses, and ultimately pose possible options for the EPA Administrator to consider with regard to whether to retain or, if appropriate, revise the Lead NAAQS. The draft Lead Staff Paper and analyses will also be made available for review by the public and CASAC. Taking into account CASAC and public comments, EPA expects to produce revised exposure and risk analyses as well as a revised draft Lead Staff Paper for public comment and CASAC review before final revisions are made in the Lead Staff Paper, to provide information to inform the decisions to be made by the EPA Administrator regarding possible retention or revision of the Lead NAAQS. The proposed NAAQS decisions will then be made available via the Federal Register for public comment and, following consideration of comments received, the EPA Administrator will promulgate final Lead NAAQS decisions via their announcement in the Federal Register.

1.3 ORGANIZATIONAL STRUCTURE AND CONTENT OF THE DOCUMENT

1.3.1 Ascertainment of Literature and General Document Format

Lists of references published since completion of the 1986 Lead AQCD/Addendum and 1990 Supplement were made available to the authors. The references were mainly selected from information data base (e.g., Pub Med) searches conducted by EPA. However, additional references have also been added (e.g., for missed or recently published papers or "in press" publications) as work has proceeded in creating the present draft document materials. As an aid in selecting pertinent new literature, the authors were also provided with a summary of issues that need to be addressed in this revised Lead AQCD. Many such issues have been identified in

the course of previous lead criteria assessments, through interactions between EPA Lead Team and Lead Work Group members, and via workshop discussions.

The general format used in this draft document is to open each new chapter (or main section) for the updated Lead AQCD with concise summary of key findings/conclusions from the previous lead criteria assessments, especially the 1986 Lead AQCD/Addendum (U.S. Environmental Protection Agency, 1986a,b) and 1990 Supplement (U.S. Environmental Protection Agency, 1990). After presentation of such background information, the remainder of each chapter or section typically provides an updated discussion of newer literature and resulting key conclusions. In some cases where no new information is available, the summary of key findings and conclusions from the previous lead criteria assessment(s) must suffice as the basis for current key conclusions. Increased emphasis is placed in the main chapters of this revised Lead AQCD on interpretative evaluation and integration of evidence pertaining to a given topic than was typical of many previous EPA air quality criteria documents, with more detailed descriptions of individual studies or other supportive information being provided in a series of accompanying annexes.

1.3.2 Organization and Content of the Document

This updated Lead AQCD critically assesses scientific information on the health and welfare effects associated with exposure to the concentrations of lead in ambient air. The document is not intended to be a detailed, exhaustive review of the literature. Rather, the cited references reflect the current state of knowledge on the most relevant issues pertinent to decisions regarding possible revision by EPA of the Lead NAAQS. Although emphasis is placed mainly on the discussion of health and welfare effects data, other scientific data is also presented and evaluated, in order to provide a better understanding of the nature, sources, distribution, and concentrations of lead in ambient air, as well as the measurement of human exposure to lead.

The focus of the selected scientific information in the text is on information published since the previous assessments of air quality criteria for lead contained in the 1986 Lead AQCD/Addendum or 1990 Supplement. Emphasis is placed on studies conducted at or near lead concentrations found in ambient air. Other studies are included if they contain unique data (e.g., the documentation of a previously unreported effect or of a mechanism for an observed effect),

or if they are multiple-concentration studies designed to characterize exposure- or dose-response relationships.

As noted earlier, key findings and conclusions from the 1986 Lead AQCD/Addendum and 1990 Supplement are typically first briefly summarized at the outset of discussion of a given topic, with appropriate reference back to the previous criteria assessment materials. Typically, important prior studies are more specifically discussed only if they are open to reinterpretation in light of newer data and/or are judged to be potentially useful in decisions on revision of the standards for lead. Generally, only information that has undergone scientific peer review and has been published (or accepted for publication) in the open literature through August, 2005 has been thus far considered in this draft criteria document. It is expected that, ultimately, the final Lead AQCD will consider new peer-reviewed studies published through December 31, 2005. Certain other unpublished analyses (e.g., de Novo analyses of recently available U.S. lead air quality data) may be considered, depending on the importance of the subject information and its pertinence to criteria development for Lead NAAQS, as determined in consultation with CASAC.

The final AQCD will consist of two volumes. Volume 1 will consist of eight chapters that comprise the main body of the revised Lead AQCD. In the first volume of this draft document, this introductory chapter (Chapter 1): (a) provides brief statements regarding the purpose of the document; (b) presents information on the legislative background and regulatory chronology of lead criteria reviews; and (c) presents an overview of the organization of the document. Chapter 2 provides information on the physics and chemistry of lead, as well as sources, emissions, transport and deposition/fate. Chapter 3 discusses environmental concentrations, dispersal patterns, and multimedia exposure pathways. Chapter 4 focuses on the modeling of multimedia exposure impacts on human internal lead burdens, especially as indexed by blood or bone lead concentrations. Then, Chapter 5 discusses toxicologic studies of lead health effects in humans, laboratory animals, and in vitro test systems; whereas Chapter 6 assesses lead-related epidemiologic (observational) studies of human population groups. Chapter 7, which will ultimately provide an integrative synthesis of key information drawn from the earlier chapters to delineate human lead exposure and health effect findings and conclusions of most importance for derivation of primary Pb NAAQS, will be prepared after CASAC review of draft Chapters 1 through 6 and will be included in the Second External Review Draft of this

Lead AQCD to be circulated for later public comment and CASAC review. Lastly, Chapter 8 deals with ecological and other environmental effects of lead as key types of welfare effects pertinent to the derivation of secondary Pb NAAQS. Several annexes containing more detailed descriptive materials supporting the interpretative evaluations highlighted in the main chapters dealing with health and vegetation/ecological effects are provided in Volume II of this revised Lead AQCD.

An Executive Summary will also be developed after CASAC review of the present First External Review Draft Lead AQCD materials and included in Volume I of the Second External Review Draft to be released for later public comment and CASAC review.

1.4 REFERENCES

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2. CHEMISTRY, SOURCES, AND TRANSPORT OF LEAD

The purpose of this chapter is to provide background information on the chemical properties of Pb that are relevant to its transport within the environment, into ecosystems and for human health considerations; to discuss the known sources of Pb in the environment; and to outline the mechanisms by which Pb is transported within the atmosphere. The chapter does not provide a comprehensive list of all sources of lead, nor does it provide emission rates or emission factors for all important source categories, since such information is available for only a limited number of sources. Rather, the chapter provides data on the chemistry, sources, and transport of lead where information is available in the peer-reviewed literature. Particle size distribution data for lead are even scarcer than total lead emissions from sources; particle size data are presented where such data are available.

2.1 PHYSICAL AND CHEMICAL PROPERTIES OF LEAD

Properties of Elemental Lead

Elemental Pb possesses an array of useful physical and chemical properties, making it among the first metals to be extracted and used by humankind. It has a relatively low melting point (327.5℃), is a soft, malleable, and ductile metal, a poor electrical conductor, and is easily cast, rolled and extruded. While sensitive to environmental acids, after exposure to environmental sulfuric acid (H_2SO_4), metallic Pb becomes impervious to corrosion due to weathering and submersion in water. This effect is due to lead sulfate ($PbSO_4$), the relatively insoluble precipitate produced by reaction of Pb with H_2SO_4, forms a protective barrier against further chemical reactions (Schweitzer, 2003). This aspect of its chemistry made Pb especially convenient for roofing, containment of corrosive liquids, and until the discovery of its adverse health effects, construction of water supply systems.

Lead is readily extracted from *galena*, a widely available sulfide mineral form of lead (PbS), by froth flotation, followed by roasting in the presence of a limited about of oxygen to form *litharge*, one of two forms of lead oxide (PbO). Elemental Pb is then isolated by reducing

PbO by way of heating in the presence of elemental carbon (coke, charcoal) (Greenwood and Earnshaw, 1987). This and other extraction and recovery processes will be discussed in greater detail, later in this chapter.

Lead alloys constitute 60% of lead used in industry (Prengaman, 2002). The major alloying elements are antimony, calcium, tin, copper, tellurium, arsenic, and silver. Selenium, sulfur, bismuth, cadmium, indium, aluminum, and strontium are also sometimes used. Lead alloys are found primarily in lead acid batteries, solder, ammunition, and cable sheathing (Prengaman, 2002). Table 2-1 provides a list of Pb alloys in use by industry.

Some of the physical properties of elemental Pb are listed in Table 2-2. The most important of these properties, when evaluating the transport routes for Pb within the atmosphere, is its boiling point. As indicated, Pb will only exist in the vapor phase at or above 1750 °C. Therefore, at ambient atmospheric temperatures, elemental Pb will deposit to surfaces or exist in the atmosphere as a component of atmospheric aerosol.

Oxidation States of Lead

Lead is the heaviest congener of carbon, and shares many properties with the other elements found in the same column of the periodic chart (silicon, germanium, and tin). As Group IV elements, these elements have four valence electrons (2 *p* and 2 *s*), allowing for both divalent and tetravalent compounds.

Due to its high atomic number (82), the valence electron orbitals of the Pb atom exist at a comparatively large distance from its nucleus. As with *s* and *p* orbitals at any quantum level, electrons in the 6*s* orbital tend to occupy space near the nucleus with greater probability than those in the 6*p* orbital. The strong attraction produced by the large Pb nucleus combined with the long distance that the 6s electrons must travel result in electron accelerations to relativistic speeds. The Theory of Relativity states that as the velocity of matter approaches the speed of light, its apparent mass increases. In this instance, the electrons in the Pb 6*s* orbital experience an increase in weight, which increases the attractive effect of the positive nuclear charge, which contracts the diameter of the Pb 6*s* orbital (Pitzer, 1979). This "relativistic effect" on valence electrons is proportional to the square of atomic number, and manifests within the Group IV elements as a distinctly increasing trend in the stability of the divalent state from Si down to Pb. In the case of Pb, the two 6*s* electrons behave as if they were chemically inert, leaving only the

Table 2-1. Lead Alloys and Their Industrial Applications

Lead Alloy	Uses
Lead-Antimony	Grids, posts, and connectors for lead-acid batteries, ammunition, cable sheathing, anodes, tank linings, pumps, valves, and heating and cooling coils
Lead-Calcium	Automotive, standby power, submarines, and specialty sealed batteries, electrowinning anodes, cable sheathing, sleeving, specialty boat keels, and lead alloy tapes
Lead-Tin	Soldering for electronics, general purposes, automobile radiators, and heat exchangers, corrosion resistant coatings on steel and copper, cable sheathing, fuses, sprinkler system alloys, foundry pattern alloys, molds, dies, punches, cores, mandrels, replication of human body parts, and filters for tube bonding
Lead-Copper	Lead sheet, pipe, cable sheathing, wire, fabricated products, tank linings, tubes for acid-mist precipitators, steam heating pipes for sulfuric acid or chromate plating baths, and lead sheathing for roofs
Lead-Silver	Anodes, high-temperature solders, insoluble anodes in the electrowinning of zinc and manganese, and soft solders
Lead-Tellurium	Pipes, sheets, shielding for nuclear reactors, and cable sheathing
Lead-Bismuth	Fuses, sprinkler system alloys, foundry pattern alloys, molds, dies, punches, cores, mandrels, solders, replication of human body parts, and filters for tube bonding
Lead-Cadmium	Fuses, sprinkler system alloys, foundry pattern alloys, molds, dies, punches, cores, mandrels, solders, replication of human body parts, and filters for tube bonding
Lead-Indium	Fuses, sprinkler system alloys, foundry pattern alloys, molds, dies, punches, cores, mandrels, solders, replication of human body parts, filters for tube bonding, and joining metals to glass
Lead-Strontium	Battery grids
Lead-Lithium	Bearings, lead-acid battery grids
Lead-Antimony-Tin	Printing, bearings, solders, slush castings, and specialty castings
Lead-Calcium-Aluminum	Negative battery grids of lead-acid batteries
Lead-Calcium-Tin	Positive grids of lead-calcium batteries, and lead anodes for electrowinning
Lead-Calcium-Silver	Zinc electrowinning
Lead-Antimony-Silver	Anodes used for the production of thin copper foil in electronics, and anodes in cathodic protection of steel pipes and structures in water
Lead-Silver-Tin	Anodes in cathodic protection of steel pipes and structures in water, and soft solders
Lead-Strontium-Tin	Anodes for copper electrowinning
Lead-Lithium-Tin	Lead-acid battery grids

Source: Prengaman (2002).

Table 2-2. Physical Properties of Elemental Lead

Physical Property	
Atomic number	82
Atomic weight	207.2
Valence electrons	$[Xe]4f^{14}5d^{10}6s^{2}6p^{2}$
Melting point	328 °C
Boiling point	1750 °C
Density	11.34 g/cm^{3}
Atomic radius	146 pm
Standard reduction potential	−0.126V
Oxidation numbers	+2, +4
Ionization Energy	715.6 kJ/mol

Source: Kotz and Purcell (1991).

two 6*p* electrons available for bonding or oxidation under ordinary conditions. For this reason, the relativistic effect is also known as the "inert pair effect." Consequently, Pb(II) is the most common oxidation state in which Pb is found in the environment (King, 1995; Claudio et al., 2003).

Lead is distinguished from other elements that are subject to relativistic effects by its preference for forming tetravalent (Pb(IV) organometallic compounds, however. In fact, it is only with rare exception that Pb(II) organometallic compounds form (Pelletier, 1995; Greenwood and Earnshaw, 1984). All simple alkyllead compounds, such as the well-known fuel additives, tetramethyllead (TML) and tetraethyllead (TEL) are composed of Pb(IV). In contrast, inorganic Pb(IV) compounds, such as PbO_2 are strong oxidants, and unstable with respect to their Pb(II) analogs. There are, overall, more than 200 known organolead compounds (Harrison, 1985).

In relation to the other Group IV metals, however, Pb forms the least stable and most reactive organometallic derivatives. This is largely due to the weak bond between lead and carbon, consistent with its large atomic size, and the influence of the relativistic effect on its valence orbitals. Specifically, the mean bond dissociation energies of the metal-carbon bonds for Group IV elements are 56.7 kcal/mol for germanium, 46.2 kcal/mol for tin, and 30.8 kcal/mol for

lead (Shapiro & Frey, 1968). Organolead compounds are thermally unstable and will decompose to metallic lead and free radicals at relatively low temperatures (Willemsen and van der Kerk, 1965). For example, TML decomposes at temperatures above 200°C, and TEL decomposes at temperatures above 110°C (King, 1995). In solution, organolead compounds decompose in the presence of UV radiation (1 hr/254 nm) and sunlight. (Gomez Ariza et al., 2000)

Tetralkyllead compounds have atmospheric residence times ranging from a few hours to a few days (Pelletier, 1995). TML and TEL react with OH in the gas-phase, following pseudo-first order kinetics, to form a variety of products that include ionic trialkyllead (TriAL), dialkyllead (DiAL) and metallic Pb. Trialkyllead is slow to react with OH and is quite persistent in the atmosphere (Hewitt and Harrison, 1986; Harrison and Laxen, 1980).

Lead Oxides, Chalcogenides, and Salts

A rich variety of inorganic Pb compounds and complex salts can be prepared in the laboratory under conditions of temperature and pressure not usually seen in the environment. Information on the many possible organic and inorganic Pb compounds can be found in the text by Greenwood and Earnshaw (1984). Several representative Pb salts and oxides are described in Tables 2-3 and 2-4. Inorganic Pb compounds that can be found in the environment are the focus of this discussion.

As explained earlier, Pb exists preferentially in its +2 oxidation state in the environment. Under aqueous acidic conditions, Pb readily oxidizes, with a strongly positive electrochemical potential ($E^0 = 1.355$ V), and a large equilibrium constant ($K = 10^{91.6}$), to form Pb(II) (Singley, 1994):

$$2Pb + O_2 + 4H^+ \rightarrow 2Pb^{2+} + 2H_2O \qquad (2\text{-}1)$$

Table 2-5 lists the various Pb compounds and salts introduced into the environment by natural processes and anthropogenic activities that will be discussed in Sections 2.2 and 2.3 of this chapter. From this list, it is clear that only a relatively limited number of salts and covalently-bound Pb compounds are of significance in the environment, i.e., sulfates ($PbSO_4$), chlorides ($PbCl_2$), carbonates ($PbCO_3$, $Pb(HCO_3)_2$), hydroxides ($Pb(OH)_2$), nitrates ($Pb(NO_3)_2$), phosphates ($PbPO_4$, $Pb(HPO_4)_2$), silicates, oxides (PbO, Pb_3O_4), and PbS. With the exception of

Table 2-3. Lead Salts: Names, Formulae, Physical Characteristics, and Uses

Category	Compound Name	Formula	Form	Uses
Lead Acetates	Anhydrous Lead Acetate	$Pb(C_2H_3O_2)_2$	White, crystalline solid	Preparing other lead salts
	Basic Lead Acetate	$2Pb(OH)_2$ $Pb(C_2H_3O_2)_2$	Heavy, white powder	Sugar analysis
	Lead Acetate Trihydrate	$Pb(C_2H_3O_2)_2$	White, monoclinic crystalline solid	Making other lead compounds, mordant for cotton dyes, water repellant, processing agent for cosmetics, perfumes, and toiletries
	Lead Tetraacetate	$Pb(C_2H_3O_2)_4$	Colorless, monoclinic crystalline solid	Oxidizing agent in organic synthesis, cleaving of a-hydroxy acids, introducing acetyl groups in organic molecules
Lead Carbonates	Lead Carbonate	$PbCO_3$	Colorless, orthorhombic crystals	Catalytic polymerization of formaldehyde, improving the bonding of polychloroprene to metals in wire-reinforced hoses, a component of high-pressure lubricating greases, and a lubricant for polyvinyl chloride
	Basic Lead Carbonate	$2PbCO_3$	White, hexagonal crystals	Ceramic glazes, a curing agent with peroxides to form polyethylene wire insulation, a color-changing component of temperature-sensitive inks, a component of lubricating grease, and a component of weighted nylon-reinforced fish nets made of polyvinyl chloride fibers
Lead Halides	Lead Fluoride	PbF_2	Colorless, orthorhombic crystals	Glass sealing disks for IR sensors, wear-resistant automotive shock absorbers, electrolytic deposition of lead, flux for brazing of aluminum and its alloys, optical glass fibers for IR transmission, and thin film batteries
	Lead Chloride	$PbCl_2$	White, orthorhombic needles	Artist's pigment, precursor of organolead compounds, seawater-activated batteries, expanding polymer mortar, flux for soldering cast iron and cast brass, sound-insulating rubber sealants, corrosion inhibitor for galvanized steel, and infrared-transmitting glasses for CO_2 lasers
	Lead Bromide	$PbBr_2$	White, orthorhombic crystals	Filler for flame-resistant polypropylene, glass optical waveguides for infrared thermometers and catalysts for producing polyesters
	Lead Iodide	PbI_2	Powdery, yellow, hexagonal crystals	Aerosols for cloud seeding, making high-contrast photographic images of laser radiation, high capacity cathodes in lithium batteries, and low-temperature thermographic copying materials
Lead Silicates	Lead Monosilicate	$3PbO•2SiO_2$	White, trigonal crystalline powder	Formulating lead-bearing glazes for ceramics, source of PbO in glass manufacturing
	Lead Bisilicate	$PbO\ 0.03Al_2O_3•1.95SiO_2$	Pale yellow powder	Ceramic glazes
	Tribasic Lead Silicate	$3PbO•SiO_2$	Reddish-yellow powder	Glass and frit production
Lead Sulfates	Tribasic Lead Sulfate	$3PbO\ PbSO_4\ H_2O$	Fine, white powder	Providing long-term heat stability to PVC, electrical insulation, activation for azodicarbonamide blowing agents for vinyl foam

Source: Carr (2002).

Table 2-4. Lead Oxides: Names, Formulae, Physical Characteristics, and Uses

Name	Formula	Form	Uses
Lead Monoxide	PbO	Reddish below 489°C, yellow at high temperatures	Pastes for the grids of lead-acid batteries, optical, electrical, and electronic glasses, glazes for fine tableware, vulcanizing agent for rubber, lead soaps used in driers as varnishes, high-temperature lubricants, neutralizing agent in organic synthesis, heat stabilizer in plastics, and starting material in the production of pigments
Lead Dioxide	PbO_2	Brownish-black crystalline powder of fine flakes	Active material of the positive plates in lead-acid batteries, oxidizing agent in the manufacture of chemicals, dyes, matches, pyrotechnics, and liquid polysulfide polymers, antifriction agent for plastic sliding bearings, ballistic modifiers in high-energy propellants, electrodes for seawater electrolysis, filters for desulfurization of waste gases, vulcanizing agents for butyl-rubber puncture-sealing layers inside tires
Lead Sesquioxide	Pb_2O_3	Amorphous, orange-yellow powder	Ballistic modifier for high-energy propellants, cathode material in lithium batteries, additive to increase the shattering force of explosives
Red Lead	Pb_3O_4	Brilliant orange-red pigment	Pigment in anticorrosion paints for steel surfaces, lead oxide pastes for tubular lead-acid batteries, ballistic modifiers for high-energy propellants, ceramic glazes for porcelain, lubricants for hot pressing metals, radiation-shielding foam coatings in clinical x-ray exposures, and rubber adhesives for roadway joints

Source: Carr (2002).

the covalently-bound sulfide and oxide, these compounds are derived from acids (or the related anions) that are common in the environment, such as sulfuric acid (H_2SO_4), nitric acid (HNO_3), carbonic acid (H_2CO_3, an acid that forms when CO_2 dissolves in water), and phosphoric acid (H_3PO_4). Lead salts, once formed, tend to be only slightly soluble in neutral solutions, but are quite soluble in the presence of acid (CRC Handbook, 1988).

Lead Coordination Chemistry, and Its Role in Biochemistry

The formation of coordinate covalent complexes represents a different class of chemical interaction from the formation of simple covalent compounds and salts. "Coordinate covalent" bonds form when anions or neutral molecules interact with metal ions in solution that are

Table 2-5. Lead Compounds Observed in the Environment

Location	Observed Pb Compounds
Minerals	PbS (*Galena*) PbO (*Litharge*, *Massicot*) Pb_3O_4 (*Minium or "Red Lead"*) $PbCO_3$ (*Cerussite*) $PbSO_4$ (*Anglesite*)
Smelting Aerosols	Pb^0, PbS $PbSO_4$, PbO, $PbSO_4$.PbO $PbCO_3$ Pb silicates
Coal Combustion Aerosols	PbS PbSe
Coal Combustion Flue Gases	Pb^0, PbO, PbO_2 (*Above 1150K*) $PbCl_2$ (*Low rank coals, above 1150K*) $PbSO_4$ (*Below 1150 K*)
Wood Combustion	$PbCO_3$
Waste Incineration Aerosols	$PbCl_2$ PbO
Soils Near Mining Operations	$PbCO_3$ $PbSO_4$ $[PbFe_6(SO_4)_4(OH)_{12}]$ $[Pb_5(PO_4)3Cl]$ $[Pb_4SO_4(CO_3)_2(OH)_3]$ $PbS\text{-}Bi_2S_3$ Pb oxides, silicates
Motor vehicle exhaust (combustion of leaded fuel)[a]	PbBrCl $PbBrCl\text{-}2NH_4Cl$ $PbBrCl\text{-}NH_4Cl$
Roadside dust[a]	$PbSO_4$, Pb^0, $PbSO_4$ $(NH_4)SO_4$, Pb_3O_4, $PbO\text{-}PbSO_4$ and $2PbCO_3\text{-}Pb(OH)_2,PbSO_4$
Other mobile sources:	
Brake wear, wheel weights	Pb^0
NASCAR vehicle emissions	Pb halides
Aircraft engine wear	Pb^0
Lawn mowers	Pb halides (*Battery leakage*)

[a]Source: Biggins and Harrison (1979, 1980).

capable of donating both of the electrons required to form a bond. These molecules (or anions) are called, "ligands," or "electron donors." Ligands possess a filled valence orbital with a geometry that allows it to overlap to a substantial degree with an empty orbital associated with the metal ion. In the case of Pb, its large atomic size is associated with several out-lying empty

atomic orbitals leading to a tendency to form a large number of coordinate covalent bonds (Claudio et al., 2003). This is suggested by the coordination number (9) of $PbCl_2$, in its crystalline form, which is able to share electrons with 9 adjacent chloride ions (Cl^-) (Douglas et al., 1983).

Molecules capable of serving as ligands for metal ions in solution take many forms. "Monodentate" ligands are molecules capable of providing 2 electrons to form a single coordinate bond, such as water (H_2O), ammonia (NH_3); "multidentate" ligands can participate in more than one coordinate bond. A common term for the binding of a metal ion by a multidentate ligand is "chelation." The chelating agent, ethylenediaminetetraacetic acid (EDTA), is a well known, hexadentate ligand, containing 6 functional groups capable of forming 6-coordinate bonds with metal ions in aqueous solution. Proteins, particularly the active sites of enzymes, contain functional groups (amino acid side-chains) that can serve as ligands for metal ions. In fact, the zinc finger proteins must form coordinate complexes with Zn^{2+} ions to stabilize their active conformation (Claudio et al., 2003).

Several types of equilibrium constants for ligand-metal interactions can be derived, depending on the property of interest. One formulation, the "binding constant (K_b)," between the free metal ion and ligands in solution, with the ligand-metal complex, is derived from the following relationship (for a neutral ligand):

$$K_b = \text{binding constant} = \frac{[ML_x^{n+1}]}{[M^{n+}][L_x]} \qquad (2\text{-}2)$$

Where: $K_{b1} = \frac{[ML^{n+1}]}{[M^{n+}][L]}$,

$K_{b2} = \frac{[ML_2^{n+}]}{[ML^{n+}][L]}$,

Etc.

Binding constants are useful, in particular, for evaluating the strength of interactions between metals and small (monodentate) ligands. The form typically used to evaluate binding between metals and proteins is the "dissociation" constant, K_d:

$$K_d = \text{dissociation constant} = \frac{[ML_{x-1}^{n+}][L]}{[ML_x^{n+}]} \quad (2\text{-}3)$$

Where: $K_{d1} = \frac{[M^{n+}][L]}{[ML^{n+}]}$,

$K_{d2} = \frac{[ML^{n+}][L]}{[ML_2^{n+}]}$,

Etc.

A variety of quantitative methods are available for establishing binding and dissociation constants for specific combinations of metals and ligands. Conversely, a simple, qualitative model for estimating the relative strength of coordinate covalent bonding between metals and ligands is the Pearson's Hard-Soft Acid-Base (HSAB) model (Douglas et al., 1983). Heavier metals, such as Pb, which have more electrons and more spatially diffuse valence orbitals, are described as "soft" (Lewis) acids. Lighter metals, with fewer electrons and more closely-spaced valence orbitals, are described as "hard" (Lewis) acids. These metals tend to preferentially bond with ligands with similar electronic properties. Hard acids tend, for example, to prefer oxygen-based ligands, i.e. "hard bases," and soft acids prefer ligands based on larger atoms, such as sulfur and selenium, i.e., "soft bases."

The HSAB concept is useful for understanding the behavior of Pb in the biological context. Lead readily forms coordinate covalent bonds with sulfur and sulfur-containing compounds, carboxylic acids and imidazoles (Claudio et al., 2003). In biological systems, Pb competes very effectively with native or homeostatic metal ions for binding with the sulphahydryl, carboxyl and imidazole side-chains comprising enzyme active sites. This competition leads to inhibition of enzyme activity, as well as the replacement of calcium in bone and, ultimately, to a substantial list of negative human health effects. The relative strength of these different interactions appears to be reasonably well-predicted by the HSAB model.

By far, the most effective biological ligands for Pb are amino acid side-chains containing sulfur and selenium. Smaller electron donors (hard bases), such as carboxylic acids that bind Pb via electrons associated with oxygen, form weaker bonds. These complexes are generally more labile, i.e., bonds form and break rapidly, thus allowing more effective competition at protein binding sites amongst metals available in solution. Example simple ligands in this case are the

amine functional group, -NH, and the thiol functional group, -SH. The amine group has a Pb binding constant on the order of 100, while the thiol group binding constant is on the order of 10^7. Example proteins in this instance are carboxypeptidase A, a zinc-binding protein, with carboxylate and histidine side-chains, and the four cysteine zinc finger consensus peptide, CP-CCC. Carboxypeptidase A has a Pb dissociation constant of approximately 10^{-4} M, versus that of the zinc finger protein, which is 3.9 X 10^{-14} M. Claudio et al. (2003) concluded, on the basis of these values, that carboxypeptidase A is unlikely to be a protein associated with Pb poisoning, while cysteine-rich proteins, including the zinc enzyme, d-aminolevulinic acid dehydratase (ALAD), the second enzyme in the heme biosynthetic pathway, are more likely targets. ALAD active site, with its Cys_3 active site, is known to be inhibited at femtomolar (10^{-15} M) concentrations of Pb in vitro.

Additional information concerning the physical aspects of Pb coordination chemistry and its role in biological systems can be gotten from the substantial review by Claudio et al. (2003). A complete discussion of the toxicology associated with exposure to Pb can be found in Chapter 5 of this document.

2.2 SOURCES OF LEAD

In this section, we summarize information on a number of major sources of lead, categorized as natural sources, stationary point sources, and mobile sources. In addition to these categories, fugitive emissions such as resuspension of lead in soil and dust can be important. Resuspension is considered a transport route and is therefore discussed in Section 2.3.

2.2.1 Natural Sources

The common sources of natural Pb include volcanoes, sea-salt spray, biogenic sources, wild forest fires, and wind-borne soil particles in rural areas with background soil concentrations. Natural sources combined contribute an estimated 19,000 metric tons of Pb to the air each year (Nriagu and Pacyna, 1988). However, there is significant variability in the Pb emissions from volcanoes and forest fires and considerable uncertainty in biogenic and sea-salt emissions of Pb (Nriagu, 1989). Table 2-6 shows the median value and the range of annual emissions worldwide for natural sources of airborne Pb.

Table 2-6. Annual, Worldwide Emissions of Lead from Natural Sources

Source	Amount Emitted: Range (thousands of metric tons/yr)	Amount Emitted: Median (thousands of metric tons/yr)
Wind-borne soil particles	0.3-7.5	3.9
Seasalt Spray	0.02-2.8	1.4
Volcanoes	0.54-6.0	3.3
Wild Forest Fires	0.06-3.8	1.9
Biogenic, continental particulates	0.02-2.5	1.3
Biogenic, continental volatiles	0.01-0.038	0.20
Biogenic marine sources	0.02-0.45	0.24
Total	0.97-23	12

Source: Nriagu (1989).

The natural lead emissions worldwide are somewhat greater than an estimated 3800 metric tons/year of lead emitted from anthropogenic stationary and mobile sources in the U.S. in the year 2000 (U.S. EPA, 2003). However, many countries around the world have much greater lead emissions than the U.S. from stationary and mobile sources, including several countries that still use leaded gasoline. Furthermore, the EPA estimate does not account for emissions of lead in resuspended soil. Harris and Davidson (2005) estimate that stationary and mobile source emissions account for only about 10% of the total lead emissions in the South Coast Air Basin of California; the remaining 90% of the emissions are from resuspended soil. The soil contains elevated lead levels because of the many decades of leaded gasoline usage. Therefore, on a worldwide basis, the anthropogenic emissions of lead are expected to be much greater than natural emissions.

There are four stable isotopes of Pb: ^{204}Pb, ^{206}Pb, ^{207}Pb, and ^{208}Pb. The last three of these isotopes are produced by decay of ^{238}U, ^{235}U, and ^{232}Th respectively. The concentrations of natural vs. anthropogenically derived Pb in environmental media are often determined through isotopic ratios. Most Pb ores exhibit ratios of $^{206}Pb/^{207}Pb$ between 0.92 and 1.20 (Erel et al., 1997). Rock released or "natural" Pb, however, generally exhibits a higher $^{206}Pb/^{207}Pb$ ratio.

Deep soil samples converge to ratios of $^{206}Pb/^{207}Pb \sim 1.21$ and $^{208}Pb/^{206}Pb \sim 2.05$ which are considerably different than the natural ratios found in adjacent bedrock (Erel et al., 1997).

Natural aerosol Pb tends to have large particle sizes (Reuer and Weiss, 2002). As a result, it deposits rapidly and has an atmospheric residence time of a few hours to ~10 days (Reuer and Weiss, 2002). The average downward flux is estimated as 0.012 mg m^{-2} yr^{-1} for natural Pb in all forms (Bindler et al., 1999).

Concentrations of lead in the air and soil have most likely been elevated by anthropogenic activities at least since the rise of the Greek and Roman societies, both of which used Pb extensively. The natural, background concentration of Pb in soil is approximately 10-15 ppm (Bindler et al. 1999; Erel et al., 1997). This is significantly higher than the adjacent bedrock but is approximately equal to concentrations found in bedrock residues such as quartz and clay (Erel et al., 1997). An estimated 3.1×10^{14} metric tons of Pb are dispersed within the continental crust (Reuer and Weiss, 2002). Of this, approximately 9.3×10^{7} metric tons of Pb are found in Pb ores. Table 2-7 lists the naturally occurring concentrations of Pb in bedrocks, ocean crusts, and continental crusts.

Table 2-7. Naturally Occurring Lead Concentrations in Major Rock Types

Lithology	Natural Lead Concentration (ppm)
Continental Crust	15.0
Oceanic Crust	0.9
Basalts, Gabbros	3.5
Limestones	5.0
Granulites	9.8
Greywackes	14.0
Gneisses, Mica Schists	22.0
Shales	22.0
Granites	32.0

Source: Reuer and Weiss (2002).

Natural Pb in surface water is derived from four different sources: biogenic material, aeolian particles, fluvial particles, and erosion (Ritson et al., 1994). About 90% of natural Pb in surface waters is in the dissolved phase (Reuer and Weiss, 2002). Organic ligands are complexed with 50-70% of this Pb with the balance found in inorganic compounds (Reuer and Weiss, 2002). Biological particles in the open ocean scavenge a significant portion of the Pb complexes, which have an estimated two-year residence time in the surface waters (Reuer and Weiss, 2002).

A naturally occurring, radioactive isotope of Pb, ^{210}Pb, is commonly studied as a tracer to determine how particles are transported through the environment. The source of ^{210}Pb is the ^{238}U decay series. In this process, gaseous ^{222}Rn is produced, which escapes from the soil and enters the atmosphere. As radon decays into ^{210}Pb, the particulate Pb deposits onto soils and surface waters all over the world. The surfaces of all soils have been exposed to atmospherically derived Pb particles (Bindler et al., 1999).

Particles of ^{210}Pb tend to be submicron with an average size of 0.53 μm AMD (Winkler et al., 1998). The mean residence time for ^{210}Pb in the air is approximately 4-5 days but has been estimated as long as 8 days with some seasonal variability (Winkler et al., 1998). The downward flux has been estimated as 136 Bq m^{-2} yr^{-1} for ^{210}Pb (Joshi et al., 1991). This results in natural, background concentrations in the soil of ≤0.1 μg/g (Bindler et al., 1999).

Atmospheric deposition is likely the largest source of ^{210}Pb to water bodies. Leaching of Pb naturally contained in host rock is a very small source to water (Toner et al., 2003). Lead-210 in surface waters is primarily in particulate form, while dissolved Pb is transported more readily (Joshi et al., 1991). Dissolved ^{210}Pb is scavenged by suspended matter (Carvalho, 1997). The residence time of dissolved ^{210}Pb is approximately 30 days although partial re-dissolution from bottom sediments probably occurs (Carvalho, 1997). One estimate found that ~56% of atmospherically derived ^{210}Pb in lakes of the Canadian Shield was retained in the sediment (Joshi et al., 1991).

Many authors have measured concentrations of ^{210}Pb in plants (including foodstuffs) and animals (including humans). Holtzman (1978) summarized these measurements. Concentrations in United States vegetation range between 30 pCi/kg and 70,000 pCi/kg for wheat and lichens respectively. The estimated human consumption of ^{210}Pb from vegetation averages 1.4 pCi/day in the United States. Overall the concentrations of ^{210}Pb in animals vary significantly depending

on the type of tissue or organ measured. However concentrations are generally higher in animals with higher rates of Pb intake.

2.2.2 Stationary Sources

Primary and Secondary Lead Smelters

Primary Pb smelting is the process by which elemental Pb is recovered from Pb ore. Lead ore is primarily in the form of galena (PbS) but can also occur as plattnerite (PbO_2), cerussite ($PbCO_3$), and anglesite ($PbSO_4$) (Reuer and Weiss, 2002). Producing elemental Pb from ore involves three processes – sintering, reduction, and refining – each with its own characteristic emissions. Primary Pb production in the United States emitted about 565 metric tons of Pb in 2000, approximately 14.7% of total anthropogenic Pb emissions in the United States (U.S. Environmental Protection Agency, 2003).

Secondary Pb smelters reclaim scrap Pb. Both the principal input to and the principal major product market of secondary smelters are lead-acid batteries. Secondary Pb production contributed 82% of total Pb production in 2003 (USGS, 2003). Secondary Pb production in the United States emitted about 422 metric tons of Pb in 2000, approximately 11.0% of total anthropogenic Pb emissions in the United States (U.S. Environmental Protection Agency, 2003).

The quantity of Pb emitted from a given facility is highly variable and depends on facility processes and meteorological conditions such as wind speed and ambient temperature. Emissions estimates are typically performed through direct measurements, mass balances, process models, inverse inferences, or emissions factors (Frey and Small, 2003).

Emissions estimates and measurements in the literature are scarce. The AP-42 program of the U.S. EPA publishes emission factors for each process in the overall smelting sequence. These emission factors are usually expressed as grams of Pb emitted per kg of Pb processed (U.S. Environmental Protection Agency, 2005). The reader is referred here for emission factors not listed below. A survey of approximately 50 European Pb smelters had mean emission factors of 0.1 grams and 0.05 grams of Pb emitted per kg of Pb processed for primary and secondary Pb smelters respectively (Baldasano et al., 1997). Measurements of emissions from the blast furnace of a primary smelter were between 1.2 and 3.8 kg Pb/hr (Bennett and Knapp, 1989). The acid-sinter at the same plant emitted between 0.4 and 8.5 kg Pb/hr (Bennett and Knapp, 1989). Emissions occur during every stage of the overall smelting process. Because the process

emissions mostly are controlled to conserve raw materials, the largest source of emissions is likely to be fugitive dust from the transport, grinding, and storage of battery scrap (Kimbrough and Suffet, 1995), which by definition is uncontrolled.

Much work has been done to determine the species of Pb emitted from the various smelting processes. The fraction of Pb in particulate matter emissions varies significantly between processes and depends on the type of furnace used. However, Pb is often the dominant element in smelter emissions. Lead can be emitted either in particulate matter or in fumes. Lead fume emissions are particularly high if Pb blast furnace bullion is transferred in an open ladle (Wang and Morris, 1995). Major components of particulate Pb emissions are PbS, $PbSO_4$, $PbSO_4$•PbO, and elemental Pb, and minor species are $PbCO_3$, PbO, Pb silicates, and PbO litharge (Batonneau et al, 2004; Harrison and Williams, 1983; Ohmsen, 2001; Sobanska et al, 1999; Rieuwarts and Farago, 1995).

The distribution of particle sizes varies depending on temperature, process, and the conditions of each facility. Ohmsen (2001) found that Pb emissions from a blast furnace tend to be less than 1 μm in size and have a smaller diameter than particulate emissions from either the sintering process or storage areas. Higher temperatures (>600 °C) in the blast furnace tend to produce emissions with finer particle sizes. Dusts from the raw materials area tend to fall between 10 and 100 μm, while dusts from the refinery tend to fall between 1 and 30 μm (Ohmsen, 2001). Sobanska et al (1999) found that just 15% of dust particles by mass emitted from a "water jacket" furnace were smaller than 10 μm and the remaining 85% fell between 10 and 100 μm. The measurements of Harrison et al. (1981) at a primary smelter found that particles derived from combustion processes were typically between 0.1 and 2 μm, but particle size measurements showed that these particles could agglomerate to more than 10 μm if they are confined to ventilation ducts. Reported sizes from primary smelting processes are shown in Table 2-8.

The concentrations of Pb in stack outlets have been measured in several cases. Measurements taken at the stack of a blast furnace at a primary smelter ranged between 3.67 and 7.32 mg/m^3 (Bennett and Knapp, 1989). Stack concentrations at the sinter plant of the same facility ranged between 4.48 and 71.0 mg/m^3 (Bennett and Knapp, 1989). Two stacks on a blast furnace at a secondary smelting facility had Pb concentrations of 0.002 and 0.0137 mg/m^3

Table 2-8. The Mass-median Aerodynamic Diameters for Particles During Various Processes at Primary Lead Smelters

Primary Smelter Process	Average Particle Size		
	Harrison et al. (1981)	Ohmsen (2001)	Bennett and Knapp (1989)
Raw Materials	—	40 μm (range = 10-100 μm)	—
Sinter	5.1 μm	range = 10-300 μm	0.91 μm, 80% of particles <10 μm
Blast Furnace	3.4 μm	90% of particles were <1 μm	1.1 μm, 88% of particles <10 μm
Copper Drosser	9.4 μm	range = 10-300 μm	—
Refinery	—	range = ~1-100 μm, mostly <20 μm	—

Note: Where there were multiple data points, geometric means were used. Data for Harrison et al. (1981) were occasionally given as >11 μm. These values were replaced with 11 μm before calculating the geometric mean. Thus, these values represent a lower limit.

Source: Harrison et al. (1981), Ohmsen (2001), Bennett and Knapp (1989).

(Sturges and Harrison, 1986). The average values of approximately 50 European smelters were 2 mg/m^3 for both primary and secondary smelters (Baldasano, et al., 1997).

The ambient air concentrations in the immediate vicinity of smelters tend to be elevated to varying degrees depending on facility operations and meteorological conditions. In the UK, an increase of 15 μg/m^3 in the local ambient air was attributed to the emissions of a single secondary Pb smelter (Sturges and Harrison, 1986). Harrison and Williams (1983) measured concentrations of 15.8 μg/m^3, 0.691-5.1 μg/m^3, and 0.151-4.54 μg/m^3 at sites 500 m, 700 m, and 1200 m from the stacks of a primary smelter respectively. Fenceline measurements at two secondary smelters located in California ranged between 0.85 and 4.0 μg/m^3 (Kimbrough and Suffet, 1995). Air concentration data measured at 50 m, 500 m, and 800 m from the plant were slightly lower but generally the same order of magnitude as the fenceline values. Ambient concentrations measured at 12 sites within several hundred meters of three secondary Pb smelters in Manitoba were elevated (Tsai, 1987). The geometric means of these samples, which were taken over three month time spans, ranged between 0.107 and 1.69 μg/m^3. Additionally,

the area was shown to be much less likely to meet the Manitoba guideline of <5 $\mu g/m^3$ for a 24-hour average when the smelters were operating than when they were not.

Non-Lead Metallurgical Processes

Emissions of Pb from non-lead smelters can be significant. Emissions from smelters, metal works, and metal refineries depend on the type of equipment used to process the metals, the concentrations of Pb in the initial material (ore, recycled material, or alloy), the type and effectiveness of pollution controls at the facility, and the temperature of operations (Pacyna, 1986). Little work has been done to speciate Pb emissions from metallurgical facilities, although Pb emissions from a primary copper-nickel smelter are primarily in the form of PbO (Barcan, 2002). The emissions of Pb from non-lead metallurgical processes are summarized in Table 2-9.

Ore Mining and Processing

Lead mining occurs in 47 countries, although primary Pb production is on the decline (Dudka and Adriano, 1997). World mine production of Pb is approximately 2.8 million metric tons per year (Wernick and Themelis, 1998). The reserve base of Pb is estimated to be about 120 million metric tons, which will sustain current rates of mine production for 43 years (Wernick and Themelis, 1998).

Mines can be a significant source of metal emissions to the atmosphere. Lead and zinc ores, which are often mined together, frequently contain high concentrations of cadmium and arsenic (Pacyna, 1986). An emission factor for Pb mines has been reported as 0.91 grams of Pb emitted to the air per kg of Pb mined (Pacyna, 1986).

Since Pb is mined in the form of galena (PbS), emissions from Pb mines tend also to be in the form of galena (Dudka and Adriano, 1997). However, other species have been detected. In mine spoils, Pb is typically galena and secondary alternation products such as plumbojarosite [$PbFe_6(SO_4)_4(OH)_{12}$] (Rieuwerts and Farago, 1995). Other Pb forms detected in the vicinity of mines are pyromorphite [$Pb_5(PO_4)_3Cl$], which has a low bioavailability, $PbCO_3$ which is formed from the weathering of galena in the soil, leadhillite [$Pb_4SO_4(CO_3)_2(OH)_2$], $PbS{\bullet}Bi_2S_3$, Pb oxides, Pb silicates, and $PbSO_4$ (Rieuwerts and Farago, 1995).

Although mining can be considered a point source to air, mine wastes can have a major widespread effect on soil and water (Riewerts and Farago, 1995). Mines produce four different

Table 2-9. The Emissions of Lead from Non-Lead Metallurgical Processes

Metallurgical Plant	Lead Emissions	Particle Sizes MMAD = Mass median aerodynamic diameter	Location	Source
Aluminum (secondary)	0.81±0.014% of PM emissions	Fine (< 2.5 µm)	Philadelphia, USA	Olmez et al. (1988)
Aluminum (secondary)	0.098±0.031% of PM emissions	Coarse (2.5-10 µm)	Philadelphia, USA	Olmez et al. (1988)
Antimony	0.17±0.04% of PM emissions	Fine (< 2.5 µm)	Philadelphia, USA	Olmez et al. (1988)
Antimony	0.11±0.02% of PM emissions	Coarse (2.5-10 µm)	Philadelphia, USA	Olmez et al. (1988)
Brass/Bronze refinery	0.01-1% of PM emissions	na	na	Lee & Von Lehmden (1973)
Brass/Bronze refinery - blast furnace	16 g/ton produced	na	na	Pacyna (1986)
Brass/Bronze refinery - crucible furnace	10 g/ton produced	na	na	Pacyna (1986)
Brass/Bronze refinery - cupola furnace	65 g/ton produced	na	na	Pacyna (1986)
Brass/Bronze refinery - reverberatory furnace	60 g/ton produced	na	na	Pacyna (1986)
Brass/Bronze refinery - rotary furnace	60 g/ton produced	na	na	Pacyna (1986)
Copper-Nickel	184 mt/yr, 21 kg/hr	1.2 µm MMAD	Copper Cliff, Ontario	Chan & Lusis (1986)
Copper-Nickel	13.4 mt/year	0.9 µm MMAD	Falconbridge, Ontario	Chan & Lusis (1986)
Copper-Nickel (primary)	0.6-1.4% of PM emissions	na	Monchegorsk, Russia	Barcan (2002)
Copper-Nickel (primary)	2.3-3.6 kg/ton produced	na	Poland	Pacyna (1986)
Copper-Nickel (primary)	3.1 kg/ton produced	na	na	Pacyna (1986)

Table 2-9 (cont'd). The Emissions of Lead from Metallurgical Processes

Metallurgical Plant	Lead Emissions	Particle Sizes MMAD = Mass median aerodynamic diameter	Location	Source
Copper Smelter - furnace	0.24-0.52 kg/hr	0.87 µm MMAD	na	Bennett & Knapp (1989)
Copper Smelter - sinter	below detection	<0.10 µm MMAD	na	Bennett & Knapp (1989)
Copper Smelter (secondary)	54-214 g/ton produced	na	na	Pacyna (1986)
Iron Ore Recovery and Ni refinery	6 mt/year	Coarse (2.5-10 µm)	Copper Cliff, Ontario	Chan & Lusis (1986)
Iron and Steel foundry	0.01-0.1% of PM emissions	na	na	Lee & Von Lehmden (1973)
Steel works - electric-arc furnace	4.1-16.3 g/ton produced	na	na	Pacyna (1986)
Zinc-Cadmium (primary)	1.2-25 kg/ton produced	na	na	Pacyna (1986)
Zinc Smelter - furnace	0.86-1.5 kg/hr	1.8-2.2 µm MMAD	na	Bennett & Knapp (1989)
Zinc Smelter - sinter	3.6-6.0 kg/hr	0.9-2.1 µm MMAD	na	Bennett & Knapp (1989)

Source: Olmez et al. (1988), Lee and Von Lehmden (1973), Pacyna (1986), Chan and Lusis (1986), Barcan (2002), Bennett and Knapp (1989).

types of large-volume waste: mine waste, which consists of overburden and barren rocks, tailings, dump heap leachate, and mine water (Dudka and Adriano, 1997). Tailings, especially, are major sources of metal contamination to soil and water (Bridge, 2004). Acid mine drainage can contain highly elevated levels of Pb, >3000 μg/L, and can contaminate vast areas (Bridge, 2004; Kurkjian et al., 2004). Soil contamination is addressed in Chapter 3.

Mining of materials other than Pb can also release Pb to the atmosphere. Zinc-copper ores, for example, contain Pb in the range of 100-100,000 ppm (Lee and Von Lehmden, 1973), and about 6.1% of all Pb in the United States is extracted from "zinc mines" (Dudka and Adriano, 1997).

In an underground gold mine, high lead-particulate concentrations were associated with blasting (Annegarn et al., 1988). These particles were primarily Pb oxides and submicron in size. A source apportionment analysis on airborne particulate matter in an underground gold mine found that the significant sources of Pb were rock dust and diesel exhaust (McDonald et al., 2003). Concentrations of airborne Pb inside the mine were measured at 0.21 $\mu g/m^3$.

Stationary External Combustion: Coal Combustion

Coal is commonly burned as a fuel for utilities, industries, and commercial and institutional facilities. Coal combustion can be a significant local source of Pb emissions as well as a considerable regional source of airborne Pb.

Coal is pulverized, fluidized, or gasified before combustion. Generally, Pb impurities will volatilize early in the combustion process although the precise rate of vaporization depends on the distribution of Pb particles in the coal and the particle sizes (Lockwood and Yousif, 2000). As Pb vapors cool they will condense, either forming individual particles or condensing on the surface of ash particles (Lockwood and Yousif, 2000; Furimsky, 2000; Clarke, 1993; Pacyna, 1986). A high surface area to volume ratio makes fine ash particles better candidates for surface sorption than coarse particles. Additionally, recondensed Pb particles tend to be fine, with an average size of 0.2 μm (Lockwood and Yousif, 2000). The fine fraction of particulate matter from coal combustion has an enrichment factor of approximately 22 (Lockwood and Yousif, 2000).

The primary contributor of Pb emissions from coal combustion is the Pb content of the coal itself. Lead is present in all coal samples in varying amounts depending on the location of

the coalfield and even the location of the coal sample within a coalfield. Generally, Pb is present in trace amounts in the form of PbS, but can also be present as pyrite and PbSe (Lockwood and Yousif, 2000; Mukherjee and Srivastava, 2005). The rank of the coal – either bituminous, subbituminous, or lignite – does not seem to correlate with the quantity of trace elements (Mukherjee and Srivastava, 2005). The age of the coal also does not seem to impact the concentration of Pb (Ghosh et al., 1987). The most important factors contributing to Pb content of uncombusted coal seems to be local environmental conditions at the time the coal formed, and the relative proportions of organic and inorganic matter (Pacyna, 1986; Ghosh et al., 1987). Globally, the concentrations of Pb in coal range between 2 and 80 ppm (Mukherjee and Srivastava, 2005). Table 2-10 lists the range of Pb concentrations measured in four different coal components.

Table 2-10. The Range of Lead Concentrations in Coal Lithotypes

Coal Lithotype	Range of Lead Concentrations (ppm)
Vitrain	0.30 – 16.17
Clarain	4.84 – 17.55
Durain	4.10 – 11.76
Fusain	3.64 – 15.60

Source: Ghosh et al. (1987).

Coal is often combined with limestone as a way to attenuate sulfur dioxide emissions. However, limestone can contain trace elements and has been shown to increase emissions of Pb by four to six times in a fluidized bed system compared to tests performed without a limestone addition (Clarke, 1993). Other measurements performed on a fluidized bed system found that increasing limestone increased particulate emissions of Pb but decreased gaseous emissions of Pb. The overall emissions of Pb (gaseous + particulate) remained relatively constant (Furimsky, 2000). Limestone had a negligible effect on pressurized fluidized bed systems although Pb emissions from gasification systems may increase with limestone additions (Clarke, 1993).

Emissions from coal combustion depend a great deal on the process conditions at a given facility. In addition to the type of boiler, conditions such as temperature, heating rate, exposure

time at elevated temperatures, and whether the environment is oxidizing or reducing can affect emissions (Pacyna, 1986). For Pb, changes in the temperature affect the size of particles, the amount of Pb in the vaporized fraction, and the species of the emissions. At combustion temperatures of 1800 K, about 0.1% of the total ash produced was vaporized (Lockwood and Yousif, 2000). At 2800 K the vaporized fraction of the ash was increased to 20%. Additionally the ratio of air to coal during combustion can have a major effect on emissions (Furimsky, 2000). In a fluidized bed system, increasing the air to coal ratio from 1.0 to 1.10 decreased the gas to solid ratio for Pb emissions from 1.5 to 0.18 (Furimsky, 2000).

Uncontrolled combustion of coal can occur – usually as natural, in-ground coal fires – and such combustion can emit Pb (Finkelman, 2004). Although these fires have local importance, they will not be discussed in detail here.

Controlled combustion is the norm for industries and utilities. The major pollution control systems are electrostatic precipitators (ESP), wet scrubbers, and baghouses. In general, pollution control systems are most effective at removing large particles and are least effective at removing submicron particles.

ESPs are highly efficient and can remove particulates with >99.9% efficiency depending on particle size, ash resistivity, flue gas temperature, and moisture content (Clarke, 1993). ESPs are used at more than 90% of coal-fired utility boilers in the United States (Senior et al., 2000). Particles that escape EPSs are typically in the range of 0.1-1.0 μm in diameter (Senior et al., 2000).

Wet scrubbers are also more than 99% efficient (Pacyna, 1986). The majority of particles that escape are <2 μm in size (Pacyna, 1986). Wet scrubbers are used less commonly than ESPs and baghouses (Senior et al., 2000).

Baghouses or fabric filters are frequently used by coal-fired utilities. As with ESPs and wet scrubbers, the collection efficiency of baghouses is a function of particle size (Senior et al. 2000). Baghouses are >99% effective with mass emissions averaging <20 mg/m^3 (Clarke, 1993).

Very little information is published regarding the actual quantity of Pb emitted from coal-fired boilers. The EPA AP-42 program publishes emission factors for typical coal-fired boilers, although using process data specific to a given facility is likely to be more accurate. Clarke (1993) reports emissions from fluidized beds. Of the processes tested, the emissions of Pb were

highest from a 0.5 m bed with a limestone sorbent, second highest with a 1.0 m bed without a limestone sorbent, and lowest with a 0.5 m bed without a limestone sorbent (Clarke, 1993). Reducing the depth of the fluidized bed by 50% decreased the emissions of trace elements by ~5-50% probably because deeper beds undergo attrition of ash (Clarke, 1993). Olmez et al. (1988) report the Pb mass fractions of particulate matter in a stack of a coal-fired power plant. For fine particles, Pb constituted 0.041 ± 0.004%. For coarse particles, Pb constituted 0.026 ± 0.002%. Coal combustion products that underwent long-range transport from the coal-fired power plants of the Midwest contributed an estimated 0.05 $\mu g/m^3$ to the ambient air in Boston (Thurston and Spengler, 1985). Table 2-11 lists the emission factors for three different types of coal, in three different types of power plants.

Table 2-11. Emission Factors of Lead for Coal Combustion in Three Different Furnaces.

Rank	Cyclone Furnace (µg/MJ)	Stoker Furnace (µg/MJ)	Pulverized Furnace (µg/MJ)
Bituminous	85	128	55
Subbituminous	103	156	66
Lignite	144	217	92

Note: All furnaces equipped with a 99% efficient ESP. Combustion of each coal type yielded ash at 10% of starting mass.

Source: Pacyna (1986).

The species of Pb emitted from coal depends on process conditions. $PbSO_4$ was found to be the dominant Pb compound in flue gas up to 1150 K (Lockwood and Yousif, 2000). Above this temperature, Pb and PbO, both in the vapor phase, dominate. As the temperature increases, the equilibrium shifts toward Pb (Lockwood and Yousif, 2000). In pulverized coal combustion at 1800K, the Pb species found in the gas phase were PbO, elemental Pb, PbCl, and $PbCl_2$ (Furimksy, 2000). The solid phase was comprised of PbO, $PbO•SiO_2$, elemental Pb, and PbO_2 (Furimksy, 2000). As the flue gas cools, the composition of Pb changes. $PbCl_2$ increases and is the main constituent of the gas phase before condensation occurs at 900K. If low rank low

chlorine coal is used, then PbO and elemental Pb will dominate the gas phase. At 1500K, $PbSO_4$ dominates the particulate phase; at 1800K PbO_2 was the predominant Pb compound in the particulate phase (Furimksy, 2000).

The emissions of Pb from coal combustion in industrial, commercial, and residential boilers are similar to the values listed above for utility boilers. Table 2-12 lists emission factors for coal combustion.

Table 2-12. The Emissions of Lead from Industrial, Commercial, and Residential Coal Combustion

Coal-fired unit	Emission factor (g/metric ton)
Industrial cyclone boiler	1.2
Industrial stoker boiler	7.7
Industrial pulverized coal boiler	4.5
Commercial/Residential boiler (stoker or hand-fired)	2.7

Note: Data for industrial boilers assuming 10% ash fraction and 85% efficient control devices.

Source: Pacyna (1986).

Stationary External Combustion: Fuel Oil Combustion

Fuel oil combustion constitutes 15% of fossil fuel energy production in the United States. (U.S. Environmental Protection Agency, 1998). As with coal, fuel oil is used to generate energy for utilities, industries, and commercial and residential boilers. The discussion below focuses on electric power utilities, which are the largest users of fuel oil.

Fuel oil is generally combusted in tangentially-fired or wall-fired boilers. Emissions of Pb from oil combustion depend on the process conditions, the amount of Pb in the oil, and the amount of sulfur in the oil (Pacyna, 1986).

The Pb concentration in the oil is the most important factor for determining the eventual emissions from combustion. The concentration of Pb in crude oil ranges between 0.001 to 0.31 ppm (Pacyna, 1986). In general, the heavier the crude, the higher the metal concentration. Refining oil removes about 10% of metals (Pacyna, 1986).

As with coal, process conditions and the presence of pollution control devices greatly affect the rate and characteristics of emissions from fuel oil combustion. Emissions from oil-fired boilers depend on the efficiency of combustion and how much deposited material has built up in the boiler (Pacyna, 1986). Additionally, poor mixing, low flame temperatures, and a short residence time in the combustion zone cause overall particulate emissions to be greater and individual particle sizes to be larger (Pacyna, 1986). Oil, which is typically atomized prior to combustion, will emit larger particles and have a higher particulate loading when atomization is done at low pressures. Conversely, high pressure atomization leads to smaller particles and lower particulate loadings (Pacyna, 1986). In general, about 90% of particulate matter mass is <2.5 μm in diameter (Olmez et al., 1988).

Many emission factors for fuel oil combustion processes are published in the AP-42 guidelines. These are of limited quality, but the reader is directed there for more information. Additional published data follow. An average emission factor for European oil-fired power plants was reported as 126 μg Pb/MJ for oil containing 1% sulfur (Pacyna, 1986). Lead emissions are higher for oils with greater sulfur contents. Olmez et al. (1988) report Pb mass fractions for two oil-fired power plants in Philadelphia. Lead was found to be 1.0% ± 0.2% and 1.8% ± 0.6% in the fine fraction in these two plants, respectively, and 0.48% ± 0.2% and 3% ± 0.4% in the coarse fraction. Lead in particulate matter at the Philadelphia plants was enriched by more than a factor of 1000 compared to the Pb concentration in the fuel oil. Lead in particulate matter for seven other oil-fired power plants was enriched by more than a factor of 100 (Olmez et al., 1988). A plant in Boston increased the ambient concentration of fine Pb aerosols by an estimated 0.05 $\mu g/m^3$ and the ambient concentration of coarse Pb aerosols by 0.003 $\mu g/m^3$ (Thurston and Spengler, 1985).

The combustion of used oil is also common. About 75% of used oil, which is generated in the transportation, construction, and industrial sectors, is burned as fuel oil (Boughton and Horvath, 2004). The Pb concentration of used oils is markedly higher than that of low-sulfur crude-based heavy fuel oils (Boughton and Horvath, 2004). Emissions from used oil combustion are estimated at approximately 30 mg of Pb from the combustion of 1 L of used oil. This is 50-100 times higher than emissions from crude-derived fuel oils.

Emission rates for industrial boilers are similar to those of utility boilers. Industrial oil-fired boilers are not usually equipped with pollution control devices. Approximately 6.4 g of Pb are emitted for 1000 L of fuel oil burned with a sulfur content of 1% (Pacyna, 1986).

Commercial and residential boilers, which are also not typically equipped with pollution control devices, have emissions of approximately 3.3 g of Pb emitted per 1000 L of fuel oil (Pacyna, 1986).

Stationary External Combustion: Wood Combustion

Wood-fired boilers are used almost exclusively by industries that produce wood or wood products. These include pulp and paper mills, lumber production facilities, and furniture manufacturers (U.S. Environmental Protection Agency, 1998). The materials used as fuel may include bark, slabs, logs, cuttings, shavings, pellets, and sawdust.

During combustion, elemental pollutants such as Pb are converted to their oxide forms. These are hydrated and later carbonated under atmospheric conditions (Demirbas, 2003a).

As with coal and oil, the largest factor affecting emissions from wood combustion is the concentration of Pb in the fuel. Lead concentrations tend to be very low for virgin wood. The median Pb concentration in 24 pine and spruce samples was 0.069 ppm (Krook et al., 2004). The concentrations of Pb in spruce, beech, oak, pine, and ailanthus are listed in Table 2-13.

Waste wood recovered from construction and demolition sites is increasingly used as fuel. Although most of this wood is untreated, some can have elevated levels of metals from surface treatment of the wood or industrial preservatives (Krook et al., 2004). Additionally, waste wood commonly contains contaminants such as metal pieces, concrete, stone, gravel, glass, and soil, which may increase metal emissions during combustion. Lead has been measured in waste wood at levels ~40 times higher than levels found in virgin wood. The median concentration of Pb in recovered waste wood in Sweden was 33 ppm (Krook et al., 2004). Lead in recovered waste wood from Germany and the Netherlands had a median value of 110 ppm.

Emissions of metals from wood are affected by process conditions. Good air-fuel mixing and high furnace temperatures keep emissions low (Demirbas, 2003a). Additionally emissions depend on whether or not the wood was combined with other fuels, the feed rate, the physical state of the wood, the stack temperature, the geometry of the boiler which can act as an inertial

Table 2-13. The Concentrations of Lead in Biomass, Char, and Ash Samples from Spruce, Beech, Oak, Pine, and of Ailanthus Trees

Wood	Biomass (ppm)	Char (ppm)	Ash (ppm)
Spruce trunk wood	0.32[a]	2.5[a]	33.2[a,b]
Beech trunk wood	0.36[a]	2.6[a]	35.0[a,b]
Oak trunk wood	0.27[a]	2.1[a]	28.4[a,b]
Pine trunk wood	n.a.	n.a.	34.9[b]
Ailanthus trunk wood	n.a.	n.a.	32.7[b]
Spruce bark	0.38[a]	3.1[a]	5.2[a], 36.2[b]
Beech bark	0.43[a]	3.3[a]	3.8[a], 40.8[b]
Oak bark	0.31[a]	2.5[a]	4.0[a], 34.0[b]
Pine bark	n.a.	n.a.	38.7[b]
Ailanthus bark	n.a.	n.a.	35.7[b]

[a] Source: Demirbas (2003a).
[b] Source: Demirbas (2003b).

particulate collector, the draft setting, and the amount of moisture in the fuel (Demirbas, 2003a; Fels et al., 1990; Pacyna, 1986).

Pollution control devices may be present with large-scale wood-fired boilers. These can greatly reduce particulate emissions. However, in a wood-burner installation in Ontario, a cyclone was found to have an efficiency of just 53% for total PM mass (Fels et al., 1990). For particles <2 μm in diameter, the concentrations downstream of the cyclone were actually greater than those upstream, probably indicating that larger particles were breaking apart during passage through the cyclone. The emissions of Pb from wood combustion are highly variable. The emission factor for wet fuel at a large-scale wood burner was 0.0006 g Pb/kg fuel (Fels et al., 1990). For dry fuel, emission factors were in the range <0.00035 to 0.0014 g Pb/kg fuel burned with an average of 0.00056 g Pb/kg fuel (Fels et al., 1990). Emissions from a wood stove and a fireplace are estimated as 0.007 g Pb and 0.0047 g Pb per kg of wood burned, respectively (Pacyna, 1986).

Emissions from the combustion of waste wood are higher than emissions from combustion of virgin wood. Although emission factors are not available, the concentration of Pb in ash is elevated above that from the combustion of virgin wood (Krook et al., 2004).

Data on particle sizes and species of emitted aerosols from wood combustion are not readily available.

Stationary Combustion Sources: Solid Waste Incineration

Incineration of municipal waste is on the decline in the United States. Historically it has been an important source of Pb emissions and locally it is still a concern in some places (Walsh et al., 2001). In New York City in the late 1960s, emissions from refuse incineration were between 602 and 827 tons per year, an appreciable fraction of the emissions from cars, which totaled ~1752 metric tons (Walsh et al., 2001).

Incinerator residue is partitioned into bottom ash, fly ash, and flue gas. Here we focus on Pb in flue gas, due to its importance in increasing airborne Pb concentrations (Chang et al., 1999). Lead in incinerator effluents is derived primarily from the noncombustible materials that end up in refuse (Pacyna, 1986). These incinerators may be equipped with pollution control devices such as cyclones, baghouses, ESPs, electrified gravel beds, and venturi scrubbers (U.S. Environmental Protection Agency, 1998).

Factors that affect the quantity of Pb emitted from incinerators include combustion temperature, the amount of Pb in the refuse, process conditions, moisture content, the addition of reactive species such as calcium, magnesium, and aluminum, and the addition of sorbents. Of all these factors, temperature seems to have the greatest impact on metal volatility (Chen and Yang, 1998). Metal volatilization is fast during the initial stages of combustion but levels off after about 15 minutes (Ho et al., 1993; Chen and Yang, 1998). When plastics only were burned, Pb volatility was at 18% at 600 °C, 61% at 800 °C, and 91% at 1000 °C (Chen and Yang, 1998). Figure 2-1 shows the percent volatility for Pb at four different combustion temperatures over 25 minutes of combustion time. Chang et al. (1999) derived the following relationship for Pb emissions from a fixed bed refuse incinerator in Taiwan:

$$\ln E(wt\%) = -3.083T^{1.257} + 3.659 \qquad (2\text{-}4)$$

where E is the weight percent of Pb in particulate emissions, and T is the combustion temperature in Kelvin.

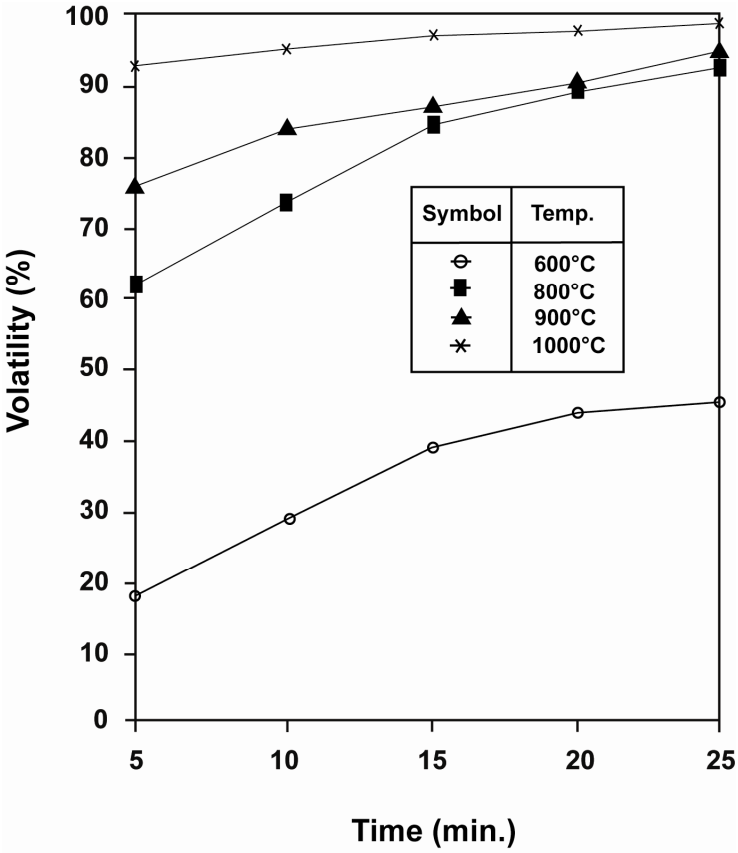

Figure 2-1. Percentage volatility of Pb during combustion of plastics at four temperatures.

Source: Chen and Yang (1998).

The amount of Pb emitted is dependent on the quantity of Pb in refuse. Typical sources of Pb include paper, inks, batteries, cans and other metal scrap, and plastics. Plastics are the most important source of Pb in municipal solid waste. Lead is used in plastic dyes and stabilizers, and plastics constitute a large portion of the waste stream (U.S. Environmental Protection Agency, 1998; Wagner and Carabello, 1997). For United States municipal solid waste, Pb concentrations vary between 110 and 1500 ppm with an average of about 330 ppm (Durlak et al., 1997). Since other countries have very different waste compositions, Pb concentrations elsewhere can vary greatly.

Additionally, process conditions can affect Pb emissions. Increasing the amount of oxygen accelerates the rate of metal volatilization but does not seem to affect the overall amount of metal volatilized (Ho et al., 1993). Similarly, Chen and Yang (1998) observed that changing

the N_2:O_2 ratio from 4:1 to 1:4 increased Pb volatility. An increase in the gas velocity can also increase Pb emissions although this is a relatively minor effect (Chang et al., 1999; Chen and Yang, 1998).

The moisture content in an incinerator can affect the behavior of Pb. At a typical temperature of 950 °C, decreasing the moisture level from 37% to 5% increased Pb in the fly ash from 54% to 58% (Durlak et al., 1997). Similarly, decreasing the relative humidity from 60% to 40% at 900 °C increased the Pb volatility from 67% to 76%, respectively (Chen and Yang, 1998). In addition to these direct effects, moisture can indirectly affect emissions by altering the combustion temperature (Durlak et al., 1997).

Additives can reduce metal emissions from incinerators. Additives such as calcium, magnesium, and aluminum react with metals and bind them. This has been shown to reduce the formation of metal particulates. Adding $Al(NO_3)_3$, for example, reduced quantities of $PbCl_2$ emitted (Ho et al., 1993). The addition of $Ca(OH)_2$ did not affect volatility at 600 °C (lower limit for combustion temperature) or at 1000 °C (upper limit for combustion temperature) (Chen and Yang, 1998). However, $Ca(OH)_2$ did appreciably limit Pb emissions at intermediate temperatures.

Sorbents can also reduce metal emissions. Sorbents function by binding metal vapors through heterogeneous chemical absorption and/or condensation before vaporized metals are able to form particulates (Ho et al., 1993). In a fluidized bed incinerator, the efficiency of metal capture with sorbents varied between 4.9% and 94.5% (Ho et al., 1993). The efficiency was dependent on temperature. Low efficiencies were observed at high and low temperatures; the optimal efficiency was observed in the intermediate range of ~600-800 °C. Limestone was shown to be a more effective sorbent than sand.

Emissions from refuse incinerators have been reported as 0.018 g of Pb emitted per kg of refuse assuming a control device with 85% efficiency (Pacyna, 1986). A source apportionment study showed that refuse incineration increased the ambient concentration of Pb by an estimated 0.008 $\mu g/m^3$ (Thurston and Spengler, 1985). This was observed after incinerators had been banned in the area probably indicating prohibited, residential refuse combustion. Lead in PM emissions has been reported between 6.9% and 8.9% with an average of 8.1% (Pacyna, 1986). Three United States incinerators had emissions in which Pb constituted 8.2 ± 1.6% of the particulate matter (Olmez et al., 1988).

Chlorine plays a critical role in determining the speciation of Pb emissions. Lead exists primarily as chlorine species (either PbCl or $PbCl_2$) (Durlak et al., 1997). However an increase in moisture content decreases the levels of free chlorine, which has the subsequent effect of shifting Pb from gaseous $PbCl_2$ to PbO in particulate form. $PbCl_{2(g)}$ is completely volatilized at 430 °C (Chen and Yang, 1998; Chang et al., 1999). Above 800 °C $PbCl_2$ slowly decomposes and $PbO_{(g)}$ and $PbCl_{(g)}$ are present in greater concentrations.

The presence of sodium can also affect speciation. Sodium has a greater affinity for binding with chlorine than Pb (Durlak et al., 1997). Thus increasing the sodium content effectively shifts the dominant Pb compound from $PbCl_2$ to PbO. Decreasing the sodium content from 6560 ppm to 4500 ppm (the average value observed in municipal solid waste) was responsible for increasing Pb in the fly ash from 35% to 60% at average moisture levels (Durlak et al., 1997). High concentrations of sodium attenuate the influence of moisture on Pb emissions.

Lead emissions tend to concentrate in the submicron size range (Chang et al., 1999; Olmez et al., 1988). Lead in the fine fraction was enriched by a factor of more than 10^5 at several United States incinerators compared with the concentration of Pb in the solid waste (Olmez et al., 1988). Lead in the coarse fraction was enriched by a factor of more than 1000.

Stationary Combustion Sources: Sewage Sludge Combustion

Sewage sludge incinerators exist at approximately 200 sites in the United States (U.S. Environmental Protection Agency, 1998). Lead can enter the sewage waste stream through car washes, galvanized material, pipe erosion, pigments, food, processed chemicals, and roofs (Krook et al., 2004). As in other combustion processes, Pb impurities vaporize during incineration and then condense.

The Pb content of dry sludge varies between 80 and 26,000 ppm with an average of 1,940 ppm (Pacyna, 1986). Sludge taken from an industrial wastewater treatment plant in Taiwan had Pb levels of 1,500 ppm (Chang et al., 1999). Prior to combustion, Pb is either bound to organic matter in sludge or is present as a carbonate (Lockwood and Yousif, 2000).

In sewage sludge incinerators, higher temperatures are associated with higher Pb emissions (Pacyna, 1986). Additionally, sewage sludge incinerators tend to be equipped with

venturi scrubbers with efficiencies of 90-99% (Pacyna, 1986). Other pollution control devices are less common.

Sorbents can be effective pollution controls. Kaolinite, in particular, was shown to reduce Pb emissions significantly (Lockwood and Yousif, 2000).

Emissions have been estimated as 0.14 g Pb emitted per kg of sludge combusted (Pacyna, 1986). The fine fraction of particulate emissions in an experimental setup was enriched with Pb by a factor of 2.5 (Lockwood and Yousif, 2000).

Stationary Combustion Sources: Scrap Tire Combustion

Waste tires are increasingly used as a fuel although uncontrolled burning as a result of accidents or illegal activity is common (U.S. Environmental Protection Agency, 1998). One analysis showed that uncontrolled combustion resulted in Pb emissions on the order of 0.47 mg Pb/kg tire for tires that had been cut into four to six pieces (Lemieux and Ryan, 1993). Emissions were lower for shredded tires, at 0.10 mg Pb/kg tire, probably because of greater oxygen transport between tire pieces. Another analysis detected trace amounts of Pb in the smoke from the combustion of tire bodies but did not detect Pb emissions when the tread was burned (Wagner and Caraballo, 1997).

Lead-acid Battery Manufacturing

Lead-acid batteries constituted 84% of Pb consumed in 2003 (USGS, 2003). Lead-acid batteries are manufactured from Pb alloy ingots and Pb oxide. Lead alloy ingots are produced by smelters, the emissions of which are characterized earlier in this chapter. Lead oxide is either produced on-site or is outsourced (U.S. Environmental Protection Agency, 1998).

Lead-acid battery manufacture consists of the following processes: grid casting or stamping, paste mixing, plate stacking, plate burning, and assembly into the battery case (U.S. Environmental Protection Agency, 1998). Each process has its own characteristic emissions of Pb. Emissions from Pb oxide manufacture tend also to be in the form of Pb oxides. These emissions are usually attenuated through a baghouse. The sites of other processes are usually equipped with baghouses or impingement wet scrubbers (U.S. Environmental Protection Agency, 1998).

Cement Manufacturing

The manufacture of Portland cement emits relatively low quantities of Pb. Trace amounts of Pb are present in the raw materials of calcium, silicon, aluminum, and iron (U.S. Environmental Protection Agency, 1998). As the raw materials are thermo-treated, most of the Pb is trapped in the resulting clinker although some is released as particulate matter (U.S. Environmental Protection Agency, 1998). Additionally, emissions result from the combustion of the coal, natural gas, or waste tires used to fire the kiln (Pacyna, 1986; U.S. Environmental Protection Agency, 1998).

Emissions are reduced significantly through the use of pollution control devices. ESPs and baghouses are both common although baghouses tend to be more effective. Lead is present in the emitted particulate matter in the range of 100 – 1000 ppm (Lee and von Lehmden, 1973). Emission factors for cement production are listed in Table 2-14.

Table 2-14. Emission Factors of Lead From Processes Used in Cement Manufacture by Control Device.

Process	Pollution Control Device		
	Multi-cyclones	ESP	Baghouse
Dry Process (total)	16.0	4.0	0.16
Kiln/cooler	12.0	3.0	0.12
Dryer/grinder	4.0	1.0	0.04
Wet Process (total)	12.0	3.0	0.12
Kiln/cooler	10.0	2.5	0.10
Dryer/grinder	2.0	0.5	0.02

Note: Units are g PB/metric ton cement.

Source: Pacyna (1986).

Glass Manufacturing

The production of leaded glass emits significant quantities of Pb. Its uses primarily include Pb crystal, cathode ray tubes for televisions, and optical glasses such as binoculars,

microscopes, and telescopes (U.S. Environmental Protection Agency, 1998). Leaded glass is composed of silica sand and Pb oxide. Lead oxide concentrations in the United States-produced leaded glass typically range between 12% and 60% but can be as high as 92% (U.S. Environmental Protection Agency, 1998).

The basic process of glass manufacturing includes blending the raw materials, melting, and forming and finishing. Lead emissions can occur during all of these processes. During blending, forming, and finishing, Pb is emitted as part of fugitive dust emissions in minor quantities (Shapilova and Alimova, 2000; U.S. Environmental Protection Agency, 1998).

The major source of emissions comes from the melting process. Emissions from melting depend mostly on the amount of Pb oxide in the raw material (Shapilova and Alimova, 2000; U.S. Environmental Protection Agency, 1998). Other factors are the type and efficiency of the furnace, the waste-gas volume, the smoke-flue length, and the efficiency of pollution control devices (Shapilova and Alimova, 2000). Electric furnaces emit significantly less Pb than gas-flame furnaces. One analysis found that the rate of Pb emissions from a gas-flame regenerative furnace was more than seven times higher than the rate of emissions from a deep tank electric furnace (Shapilova and Alimova, 2000). Baghouses are the most efficient pollution control device for glass manufacturing operations (U.S. Environmental Protection Agency, 1998). Wet scrubbers are relatively ineffective, and ESPs are between 80% and 90% effective (U.S. Environmental Protection Agency, 1998). Rates of Pb emissions from several types of furnaces are listed in Table 2-15.

Ammunition Production and Shooting Ranges

In 2003, 48,800 metric tons of Pb were consumed in the United States for the production of ammunition (USGS, 2003). Additionally, some Pb is used to produce Pb azide or Pb styphnate, which is a detonating agent. Small arms manufacturing plants are likely emitters of Pb although the actual quantity is unknown.

Shooting ranges, both outdoor and indoor, may have a local impact on airborne Pb concentrations. Lead is emitted from cast Pb bullets and lead-based primers (Gulson et al., 2002). The propellants contain <2 ppm and seem to have a negligible effect on air concentrations. A 97% reduction in the air Pb concentrations was observed when Cu-jacketed bullets replaced cast Pb bullets (Gulson et al., 2002). In comparing the Pb exposure of

Table 2-15. Rate of Lead Compound Emissions from Glass-Melting Furnaces

Equipment	Product	Lead Compound Emissions g/sec)
Electric tank furnace with gas-heated working zone[a]	Glass with 16% PbO	0.134
Electric tank furnace with gas-heated working zone	Glass with 16% PbO	0.002
Gas-flame potter furnace[a]	Glass with 16% PbO	0.004
Slag-lining electric furnace with gas-heated working zone	Glass with 64.5% PbO	0.004

[a] Fitted with a "cassette pulse filter" designed specifically to capture particulate emissions from small-sized, glass-melting furnaces.

Source: Shapilova and Alimova (2000).

personnel, there seems to be little difference between indoor and outdoor firing ranges (Gulson et al., 2002). One study found that soil Pb concentrations at an outdoor firing range were elevated by up to 2600 times background concentrations, indicating significant atmospheric deposition (DeShields et al., 1998).

An additional source of Pb emissions may be explosive ordnance disposal (EOD) (U.S. Environmental Protection Agency, 1998). Emissions from EOD are either from the combustion or detonation of the propellant and primer material or from nonenergetic wastes such as containers and other wastes associated with the propellant (U.S. Environmental Protection Agency, 1998).

Demolition

A study of Pb dust-fall during the demolition and debris removal of urban row houses found that Pb was released in very large quantities (Farfel et al., 2003). Many of the row houses demolished at three sites in Baltimore, MD contained lead-based paint in addition to being near sites with elevated levels of Pb in street dust (~700 ppm), sidewalk dust (~2000 ppm), and residential entryway mat dust (~750 ppm). The results of the study showed that dust fall within 10 m of the demolition sites was much higher than baseline measurements and was highly enriched with Pb (Farfel et al., 2003). The geometric mean Pb dust fall rate increased to 410 μg

Pb/m^2/hr during demolition and to 61 µg Pb/m^2/hr during debris removal. The baseline rate is just 10 µg Pb/m^2/hr. The Pb concentration in dust fall was 2600 ppm during demolition, 1500 ppm during debris removal, and 950 ppm at baseline (Farfel et al., 2003).

Other Stationary Sources of Lead Emissions

There are additional stationary sources of Pb emissions that have not been mentioned above. Each of these sources are relatively small, but may be an important local source. Previously unmentioned Pb sources include: medical waste incineration, hazardous waste incineration, drum and barrel reclamation, crematories, pulp and paper mills, pigment production, Pb cable coating production, frit manufacturing, ceramics and glaze production, type metal production, pipe and sheet Pb production, abrasive grain processing, solder manufacturing, electroplating, resin stabilizer production, asphalt concrete production, paint application, and rubber production.

2.2.3 Mobile Sources

Automotive Sources of Lead Emissions

Lead is used to manufacture many components in on-road vehicles including the battery, bearings, paint primers, corrosion-resistant gas tanks, and some plastic and ceramic electrical components (U.S. Environmental Protection Agency, 1998). The major sources of Pb emissions—fuel combustion and vehicle wear—are considered below.

Emissions from Combustion of Unleaded Gasoline

Although its phase out began in 1975, Pb was still added to gasoline in the United States as an anti-knock additive at the time of the last Criteria Document. The United States completed its phase out of Pb additives in 1990, and airborne concentrations have fallen dramatically nationwide. This is considered one of the great successes for public and environmental health (Nriagu, 1990). Airborne concentrations in the United States fell an average of 94% between 1983 and 2002 and 57% between 1993 and 2002 (U.S. Environmental Protection Agency, 2003).

Most countries have made a similar move away from leaded fuel, but a few continue the practice of adding tetraethyl Pb to automotive gasoline. Worldwide Pb consumption for gasoline peaked in the 1970s at just under 400,000 metric tons, but by 1993, this value fell to about

70,000 metric tons (Socolow and Thomas, 1997). Leaded gasoline was the largest source of air emissions throughout the 1970s and 1980s (Socolow and Thomas, 1997). In Pakistan, a country that continues to use leaded fuel, the airborne concentrations in the urban center of Karachi range between 2.0 and 19 μg Pb/m^3 (Parekh et al., 2002). This is 2 to 3 orders of magnitude higher than typical urban concentrations in the United States.

In the absence of tetraethyl Pb additives, Pb is emitted from automobiles as a trace element in particulate matter. Metals enter the vehicle in trace amounts, naturally occurring in gasoline. The amount of particulate matter that is emitted from the car depends on a number of variables including the ambient temperature, the cruising speed, the amount of stop-and-go activity, the type of catalyst, the fuel quality, the phase of driving, and the age, size, maintenance level, and engine type of the vehicle.

The amount of Pb that naturally occurs in gasoline is approximately 0.00005 g/L (Harris and Davidson, 2005). An estimated 30-40% of this Pb deposits in the engine and exhaust system; the balance is emitted (Hutzicker et al., 1975; Loranger and Zayed, 1994).

Particulate matter emissions have been shown to be higher in older vehicles than in newer vehicles (Gillies et al., 2001; Cadle et al., 1999). Gillies et al. (2001) compared emission factors from several studies, and found that emission factors from car models between the years 1964 and 1983 had emission factors for PM that were about an order of magnitude higher than models from the 1990s. This was true even of catalyst-equipped vehicles. Similarly, Cadle et al. (1999) tested 195 cars with model years between 1971 and 1996. Their results, which are listed in Table 2-16, show an increase in emission rates with automobile age.

Vehicles that have visible tailpipe emissions are known as "smokers." The emissions of almost all pollutants are elevated from smoking vehicles compared to their non-smoking counterparts. Emission rates of Pb from smokers are an order of magnitude higher than typical cars manufactured in the 1990s, as shown in Table 2-16. Interestingly, another study found that smoking and other high-emitting vehicles emitted more Pb after undergoing repair than before (Cadle et al., 1997). The emission rate of Pb before repair had an average value of 0.029 mg/mi with a standard deviation of 0.047 mg/mi. After repair, the emission rate for Pb increased to 0.161 mg/mi with a standard deviation of 0.346 mg/mi. The authors explain this surprising result by suggesting that either changes in combustion conditions caused elemental deposits from the

Table 2-16. Emission Factors of Lead for Automobiles with Model Years Between 1971 and 1996

Vehicle Category	Emission Factors in mg/mile	
	Summer	Winter
1991-1996	0.003	0.019
1986-1990	0.027	0.019
1981-1985	0.006	0.103
1971-1980	0.043	0.222
Smokers	0.035	0.282
Diesel	0.15	0.142

Note: "Diesel" denotes diesel automobiles, "Smokers" denotes automobiles with visible emissions.

Source: Cadle et al. (1999).

engine and exhaust system to be released, or particulate matter deposited during repair and testing was not removed before emissions testing (Cadle et al., 1997).

Table 2-16 also shows the effect of the ambient temperature on emission rates of Pb. Emissions tend to be higher during cold months than during warm months (Cadle et al., 1999).

The rate of emissions is largely dependent on the phase of driving. The Federal Test Procedure analyzes three phases: cold start, hot stabilized, and hot start, the results of which are shown in Table 2-17. Driving cycles that are not included are the highway fuel economy test, and a high speed, high load cycle known as US06 (Cadle et al., 1999). Emissions were significantly higher during cold start than during the hot stabilized and hot start phases.

Despite the large variability in Pb emissions, several studies describe average on-road emission factors for a typical fleet. Sternbeck et al. (2002) measured metal concentrations in two tunnels in Gothenburg, Sweden. The emission factors subsequently derived were 0.036 ± 0.0077 mg/km per vehicle and 0.035 ± 0.014 mg/km per vehicle for the two tunnels, respectively. Another tunnel study was performed on a fleet comprised of 97.4% light-duty vehicles and 2.6% heavy-duty vehicles in the Sepulveda Tunnel in California (Gillies et al., 2001). The emission factors for Pb were 0.08 mg/km per vehicle and 0.03 mg/km per vehicle in the PM10 and PM2.5 fractions respectively. Lough et al. (2005) analyzed emissions from on-road vehicles in two tunnels in Milwaukee, Wisconsin. Trucks constituted between 1.5% and

Table 2-17. Emission Factors of Lead for Automobiles with Model Years Between 1971 and 1996

Vehicle Category	Summer Emission Factors in mg/mile		
	Cold Start	Hot Stabilized	Hot Start
1991-1996	0.005	0.002	0.002
1986-1990	0.041	0.020	0.031
1981-1985	0.016	0.002	0.006
1971-1980	0.112	0.015	0.044
Smokers	0.116	0.010	0.031
Diesel	0.190	0.048	0.313

Source: Cadle et al. (1999).

9.4% of the vehicles, with the balance comprised of passenger cars. Lead emission rates were on the order of 0.01 mg/km per vehicle and 0.1 mg/km per vehicle in the summer and winter respectively. Cadle et al. (1999) analyzed 195 in-use, light-duty vehicles using two dynamometers. Their results are shown in Tables 2-15 and 2-16. A test on noncatalyst-equipped, light-duty vehicles found that Pb constituted about 0.03% of the fine particle mass emitted from these vehicles (Kleeman et al., 2000).

Vehicle-derived Pb seems to have a bimodal distribution. The submicron mode is likely the product of combustion or high temperatures, and therefore probably came from the tailpipe (Lough et al., 2005; Harrison et al., 2003; Abu-Allaban et al., 2003). The coarse mode, with an approximate size range of 1.0 to 18 μm in diameter, is likely a product of physical processes such as road dust resuspension and tire or brake wear (Lough et al., 2005; Abu-Allaban et al., 2003). More than 80% of the airborne Pb particles near a roadway were <$PM_{2.5}$ (Harrison et al., 2003).

Emissions from Combustion of Diesel Fuel

In on-road studies of a typical fleet, as in tunnel studies, the relative contributions of diesel fuel and gasoline are difficult to separate.

Emissions of particulate matter from diesel vehicles are highly dependent on the mode of operation (Shah et al., 2004). Emission rates are much higher in simulated congested traffic situations than at cruise or highway speed conditions (Shah et al., 2004).

Extensive profiles of diesel emissions were developed by Lowenthal et al. (1994). Their results for Pb are summarized in Table 2-18.

Table 2-18. The Concentration of Lead in Particulate Matter Emissions and Emissions Factors for Lead from Buses and Trucks Fueled with Diesel No. 2 and Jet A Fuel

Fuel and Vehicle Type	Concentration of Pb in PM (%)	Uncertainty (%)	Emission Factor (mg/km)	Uncertainty (mg/km)
Truck, Diesel No. 2	0.0007	0.0028	0.0053	0.0187
Truck and Bus, Diesel No. 2	0.0006	0.0025	0.0045	0.0188
Truck and Bus, Jet A	0.0010	0.0055	0.0050	0.0214
Bus, Jet A and Diesel No. 2 with particulate trap	0.0009	0.0052	0.0016	0.0100
Bus, Jet A with particulate trap	0.0028	0.0132	0.0018	0.0085
Phoenix PM_{10} study	0.0147	0.0294	n.a.	n.a.

The results of Chow et al. (1991) on heavy-duty particulate emissions in Phoenix are listed in the last row for comparison.

Source: Lowenthal et al. (1994).

Particulate matter from diesel vehicles tends to be smaller than $PM_{2.5}$ (Gillies et al., 2001; Kleeman et al., 2000). The peak of the particle mass distribution appears to be around 0.1 µm (Kleeman et al., 2000). Although no data were available specifically for Pb, such small particle sizes would be consistent with expectations from high-temperature processes.

Emissions from Vehicle Wear

Vehicle wear and loss of Pb wheel weights are considered as sources of roadside Pb contamination. Brake wear, in particular may emit significant quantities of Pb in particulate matter. Harrison et al. (2003) note that Pb is poorly correlated with emissions of NO_x, which is

emitted from tailpipes. These authors suggest that brake wear contributes the additional quantities of Pb observed in ambient air. Sternbeck et al. (2002) compare emission factors derived in other studies. Estimates of Pb emissions from brake pads in Sweden were just under 200 µg/km per vehicle (Sternbeck et al., 2002). This is an order of magnitude higher than the tailpipe emissions measured by Cadle et al. (1999).

Up to 35% of brake pad mass loss is emitted as airborne particulate matter (Garg et al., 2000). One study that analyzed particulate emissions from seven different brake pad formulations found that only one type of brake pad described as "potassium titanate, aramid, and copper fiber" emitted particulate matter with a measurable Pb fraction (Garg et al., 2000).

A joint study in Reno, NV and Durham/Research Triangle Park, NC found that the dominant contributors to particulate matter were resuspended road dust and tailpipe emissions (Abu-Allaban et al., 2003). However, brake wear was a significant source of particulate matter in places where strong braking occurred, such as at freeway exits (Abu-Allaban et al., 2003).

Particulate matter emissions from brake pads were primarily in the fine fraction. Eighty-six percent and 63% of airborne PM was smaller than 10 and 2.5 µm, respectively (Garg et al., 2000). It is expected that Pb particles from mechanical processes such as brake wear would be in the coarse fraction. However, smaller particles may be observed if Pb is vaporized from hot brake surfaces (Harrison et al., 2003; Lough et al., 2005).

Lead weights used to balance vehicle wheels may pose an additional threat to roadside concentrations of Pb. In Albuquerque, NM deposition of Pb wheel weights was estimated to be between 50 and 70 kg/km per year (Root, 2000). Wheel weights are 95% Pb, 5% antimony, and typically weigh between 7 and 113 grams. These wheel weights can become dislodged during quick stops. Although deposited pieces of wheel weights are quite large, Pb is very malleable and can be worn away into respirable particles by being run over by vehicles (Root, 2000).

Emissions from Racing Vehicles

Vehicles used in racing (including cars, trucks, and boats) are not regulated by the EPA according to the Clean Air Act, and can therefore use alkyl-lead additives to boost octane. Data on Pb levels in racing fuel and rates of Pb emissions are scarce. The U.S. Department of Energy stopped tracking information on the production of leaded gasoline for non-aviation use in 1990 (U.S. Environmental Protection Agency, 2002). However, the National Motor Sports Council

reports that approximately 100,000 gallons of leaded gasoline were used by National Association for Stock Car Automobile Racing (NASCAR) vehicles in 1998 (U.S. Environmental Protection Agency, 2002).

As was the case with on-road emissions during the time of universal leaded gasoline use, the combustion of racing fuel likely elevates airborne Pb concentrations in the nearby area. This may pose a serious health risk to some subpopulations such as residents living in the vicinity of racetracks, fuel attendants, racing crew and staff, and spectators.

The EPA has formed a voluntary partnership with NASCAR with the goal of permanently removing alkyl-Pb from racing fuels used in the Busch, Winston Cup, and Craftsman Truck Series (U.S. Environmental Protection Agency, 2002).

Emissions from the combustion of leaded fuel are generally in the form of submicron particles of inorganic Pb halides.

In addition to racing vehicles and piston engine aircraft, legally permitted uses of leaded fuel include construction machinery, agricultural equipment, logging equipment, industrial and light commercial equipment, airport service equipment, lawn and garden equipment, and recreation equipment including boats, ATVs, jet skis, snowmobiles, etc., (U.S. Environmental Protection Agency, 2000). Given the relative unavailability of leaded fuel, it is unlikely that it is commonly used for any of these purposes other than racing vehicles.

Aircraft

Piston-engine aircraft use leaded fuel. Aviation fuel or avgas contains between 0.1 and 1.0 g of tetraethyllead additives per liter. About 32.7% of general aviation aircraft use avgas, the remainder use jet fuel, which does not contain Pb additives (Harris and Davidson, 2005). The overall fraction of aviation fuel containing Pb additives is unknown.

In the South Coast Air Basin of California, emissions of Pb from general aviation aircraft was estimated as 634 ± 110 kg/year (Harris and Davidson, 2005). This corresponds to 0.54 grams of Pb released per flight. Approximately 267 kg of the total was emitted below the mixing height in 2001, which could be a local source of Pb exposure.

Commercial jet aircraft do not use leaded fuel. However, they are also likely sources of Pb emissions. In-flight sampling of contrails from a DC-8 and a 757 showed that metals constituted more than 11% and 5.2% of particulate matter, respectively (Twohy and Gandrud,

1998). This is a lower limit for the fraction of metals in emissions since almost half of the particles in contrails are from the ambient air (Twohy and Gandrud, 1998).

No known estimates have been made of the quantity of Pb in commercial aircraft emissions. However, the dominant metals seem to be Fe, Cr, and Ni (Kärcher, 1999). These are the primary components of stainless steel and indicate that engine erosion is a significant source of metal emissions (Kärcher, 1999).

Metal particles in contrails have two modes. One is submicron with an average diameter of about 0.36 μm (Kärcher, 1999; Twohy and Gandrud, 1998). The larger mode is ~1 μm in diameter and has a morphology that suggests mechanical generation (Kärcher, 1999).

Lawn-care Equipment

A life cycle assessment used to compare gasoline-, electricity-, and battery-powered lawn mowers found that electricity-powered mowers had the fewest overall emissions over its lifetime (Sivaraman and Lindner, 2004).

Battery powered mowers are fitted with a lead-acid battery. The total amount of Pb released to the environment from the battery over its lifetime is approximately 0.052 kg Pb which includes consideration of raw material extraction and refining, energy production, Pb mining and refining, battery manufacture, and battery recycling (Sivaraman and Lindner, 2004).

Electricity-powered lawn mowers presumably emit less PM and Pb than gasoline-powered mowers. This is a reasonable assumption since utility generation plants tend to be fitted with pollution control devices and internal combustion engines of gasoline-powered mowers do not.

Other Mobile Sources of Lead Emissions

Lead emissions are associated with the combustion of any fossil fuels. Thus, any of the following may be additional mobile sources of Pb emissions that are not addressed above: construction equipment, off-road recreational vehicles, generators, marine vessels, locomotives, agricultural equipment, logging equipment, and lawn and garden equipment. However, detailed data on these sources are not readily available.

2.3 TRANSPORT WITHIN THE ENVIRONMENT

2.3.1 Atmospheric Transport of Lead Particles

Atmospheric Dispersion

The atmosphere is the major environmental transport pathway for anthropogenic lead (Reuer and Weiss, 2002).

Airborne lead tends to be in the form of submicron aerosols (Davidson and Rabinowitz, 1992; Davidson and Osborn, 1986; Harrison, 1986; Lin et al., 1993). The mass median diameter averaged for several studies is 0.55 μm (Milford and Davidson, 1985). A study performed in 1991, after leaded gasoline was no longer the predominant source of lead in the atmosphere, showed a bimodal distribution for lead particles with the larger peak in the fine fraction (Lin et al., 1993). The mass median diameter for lead samples was 0.38 ± 0.06 μm in the fine fraction and 8.3 ± 0.6 μm in the coarse fraction. Since small particles are much slower to deposit than larger particles, lead can be transported great distances in the atmosphere. Detectable quantities of lead have been found even in the most remote places on earth. Because much of the airborne lead is generally associated with fine particles, atmospheric dispersion models used for gaseous pollutants can be applied to estimate atmospheric flows of lead under certain conditions. Use of such dispersion models is more accurate for submicron lead emitted from stacks than it is for larger particles resulting from fugitive emissions, such as resuspended soil particles.

The airborne concentration of a species emitted from a point source is frequently described with a Gaussian distribution. This simple description holds true only when turbulence is stationary and homogeneous. However, the Gaussian model can be modified to account for more complex atmospheric conditions. For a thorough discussion of assorted Gaussian plume models and parameters, the reader is directed toward the work of Seinfeld and Pandis (1998). Gaussian models are in general reasonably accurate for small-scale work – within approximately 100 km of the source.

The rate and direction of dispersion are dependent both on pollutant characteristics and meteorological conditions. Important meteorological factors include windspeed, surface roughness, inversion frequency, inversion duration, and the temperature.

A Gaussian dispersion model (EMITEA-AIR) was applied to theoretical primary and secondary lead smelters in Europe (Baldasano et al., 1997). This model accounts for plume rise as well as interactions between the plume and terrain. Two sites were modeled. Conditions in

Copenhagen, Denmark included flat terrain, dominant strong winds, neutral or stable turbulence, and an annual mean temperature of 10°C. Conditions in Catalunya, Spain had a complex terrain, weak winds, unstable turbulence, and an annual mean temperature of 15°C.

The results of these modeling efforts showed that airborne concentrations of lead were both lower and more symmetric surrounding the Copenhagen site than surrounding the Catalunya site (Baldasano et al., 1997). Concentrations at the Copenhagen site had a maximum value of 0.004 $\mu g/m^3$. Concentrations at the Catalunya site ranged between 0.065 and 0.3 $\mu g/m^3$. The prevalence of calm winds and the complex terrain were the most important factors contributing to high lead concentrations surrounding the Catalunya smelter.

Modeling efforts for an abandoned battery recycling facility using the EPA Industrial Source Complex Short Term (ISCST) model, based on Gaussian equations, showed good agreement with measured concentrations (Small et al., 1995). Model predictions at three sites at distances between 240 and 310 m from the stack were between 3.8 and 4.4 $\mu g/m^3$ while measured concentrations taken when the plant was in full operation had averages between 4.1 and 5.2 $\mu g/m^3$.

For long-range transport modeling, Lagrangian trajectory or Eulerian grid models are commonly employed. These models determine how a parcel of air moves relative to the moving fluid and a fixed coordinate system, respectively.

Two Lagrangian experiments were performed in the Azores in the northern Atlantic (Véron and Church, 1997). Retrospective air mass trajectories based on the hybrid single-particle Lagrangian integrated trajectory (HY-SPLIT) model found that air masses enriched with lead had been over continental regions ten days prior to testing. This is consistent with current understanding that most lead emissions are from sources on continents, not from oceanic sources. Airborne lead at this remote location was transported from several different countries (Véron and Church, 1997).

Similarly, backward air mass trajectories estimated for Greenland showed that the highest air concentrations of metals were in air parcels that had been over continental regions five days earlier (Davidson et al., 1993). The model used in this study employed a constant acceleration formulation of the trajectory equations, and encompassed air parcel movements affected by terrain and meteorology. The air masses with the highest metal concentrations were traced back

to polluted regions including the Arctic Basin, eastern North America, and Western Europe (Davidson et al., 1993).

A numerical model that combined weather system modeling with three-dimensional Lagrangian transport and diffusion modeling was used to determine the foreign contributions of lead to airborne concentrations in Israel (Erel et al., 2002). These predictions in conjunction with isotopic measurements indicated that Israel received significant amounts of lead from Egypt, North Africa, the United Arab Emirates, Jordan, Turkey, and Eastern Europe (Erel et al., 2002).

Historical Records of Atmospheric Lead Transport and Deposition

An important field of research involves analyzing natural records of lead deposited from the atmosphere. Lead concentrations are measured in media such as soil, sediments, ocean water, peat bogs, plants, snowpacks, or ice cores. Based on concentrations, ratios to other pollutants, or isotopic compositions, an airborne concentration is back calculated and in some cases the major emitters can be identified. Sediments can provide records dating back several million years, peat bogs can reach back to the late glacial period (~15000 years ago), corals and trees can record up to several hundred years, and lichens and mosses can provide recent deposition data (Weiss et al., 1999). Additionally, some applications can yield data showing variation with seasons or climate. These methods have been used to monitor both short and long-range transport. For a comprehensive look at natural historical records, the reader is referred to the review articles by Weiss et al. (1999), Boutron et al. (1994), and Garty (2001).

2.3.2 Deposition of Airborne Particles

Deposition, both dry and wet, is the major removal mechanism for atmospheric pollutants. Here we focus on deposition data published specifically for lead aerosols, although the literature on particle deposition is extensive.

Dry Deposition

Dry deposition is the process by which pollutants are removed from the atmosphere in the absence of precipitation. The downward flux, -F, is characterized by:

$$-F = V_d C \qquad (2\text{-}5)$$

where C is the airborne concentration in $\mu g/m^3$ and V_d is the deposition velocity in m/second. The deposition velocity is an empirical quantity defined by Equation 2-1 as the ratio of F to C with units of m/s. It should be noted that both the airborne concentration and the deposition velocity are dependent on vertical height.

The physical factors governing dry deposition are often described in a manner analogous to electronic resistances (Davidson and Wu, 1990). The parameters of aerodynamic resistance, boundary layer resistance, and surface resistance run in parallel with sedimentation resistance or gravity. The relative importance of each of these resistances varies with particle size and meteorological conditions (Wu et al., 1992a).

The size of depositing particles is arguably the most important factor affecting deposition rates. For very small particles, Brownian motion is the dominant mechanism that transports particles through the viscous sublayer that borders surfaces (Nicholson, 1988a). For large particles, sedimentation is the most important process governing particle deposition. For intermediate particles impaction and interception largely determine deposition rates. The deposition velocity has the most uncertainty for these intermediate sized particles (Nicholson, 1988a). Although most of the airborne lead mass is associated with submicron particles, only about 0.5% of the lead particle mass undergoing dry deposition in Chicago were smaller than 2.5 μm in diameter (Lin et al., 1993). Additionally, more than 90% of lead particle mass that undergoes dry deposition is in an insoluble chemical form (Gatz and Chu, 1986).

Deposition velocities for lead are in the range of 0.05-1.3 cm/s. Table 2-19 is a compilation of data from the literature. Figure 2-2 shows the variation of deposition velocity for lead with particle size.

Wet Deposition

Wet deposition is the process by which airborne pollutants are scavenged by precipitation and removed from the atmosphere. The flux of a depositing species can be defined through the following equation:

$$F=V_pC_p \qquad (2\text{-}6)$$

where V_p is the rate of precipitation in cm/s and C_p is the concentration of the chemical species in the precipitation (μg/L) (Miller and Friedland, 1994).

Table 2-19. Deposition Velocities for Lead Particles. Data taken from Davidson and Rabinowitz (1992), Rojas et al. (1993), Friedlander et al. (1986), Davidson and Wu (1990), Davidson et al. (1985), and El-Shobokshy (1985).

Vd (cm/s)	MMAD (μm)	Surface	Other	Author	
0.26	all	water		Davidson & Rabinowitz (1992)	
0.56	all	orchard grass		Davidson & Rabinowitz (1992)	
0.06 ± 0.02	all		model of Rojas et al.(1993)	Rojas et al. (1993)	
0.06 ± 0.02	all		model of Slinn & Slinn (1980)	Rojas et al. (1993)	
0.09 ± 0.03	all		model of Williams (1982)	Rojas et al. (1993)	
0.26	all	all	mass balance model	Friedlander et al. (1986)	
0.14 ± 0.13	10% >4	teflon plates		Davidson and Wu (1990)	
0.15 ± 0.07	0.87	teflon plates		Davidson et al. (1985)	
0.41	0.68	water		Davidson and Wu (1990)	(taken from Dedeurwaerder et al. (1983)
0.43	0.75	water		Davidson and Wu (1990)	(taken from Dedeurwaerder et al. (1983)
0.19	0.70	land		Davidson and Wu (1990)	(taken from Dedeurwaerder et al. (1983)
0.33 ± 0.03		alfalfa + oil	stable conditions	El-Shobokshy (1985)	
0.31 ± 0.02		alfalfa + oil	unstable conditions	El-Shobokshy (1985)	
0.37 ± 0.04		grass + oil	stable conditions	El-Shobokshy (1985)	
0.31 ± 0.02		grass + oil	unstable conditions	El-Shobokshy (1985)	
0.28 ± 0.05		soil	stable conditions	El-Shobokshy (1985)	
0.34 ± 0.05		soil	unstable conditions	El-Shobokshy (1985)	
0.9 ± 0.3	0.79	beech canopy	throughfall	Davidson and Wu (1990)	(taken from Hofken et al. (1983)
1.3 ± 0.5	0.79	spruce canopy	throughfall	Davidson and Wu (1990)	(taken from Hofken et al. (1983)
0.05	0.5	polyethylene petri dish		Davidson and Wu (1990)	(taken from Lindberg and Harriss (1981)
0.005	0.5	oak	foliar extraction	Davidson and Wu (1990)	(taken from Lindberg and Harriss (1981)
0.06 ±0.01	0.5	polyethylene petri dish		Davidson and Wu (1990)	(taken from Lindberg and Harriss (1981)
0.46		filter paper		Davidson and Wu (1990)	(taken from Pattenden et al. (1982)
0.06	0.3	bucket		Davidson and Wu (1990)	(taken from Rohbock (1982)
0.13	82%<1	water	aerometric mass balance	Davidson and Wu (1990)	(taken from Sievering et al. (1979)

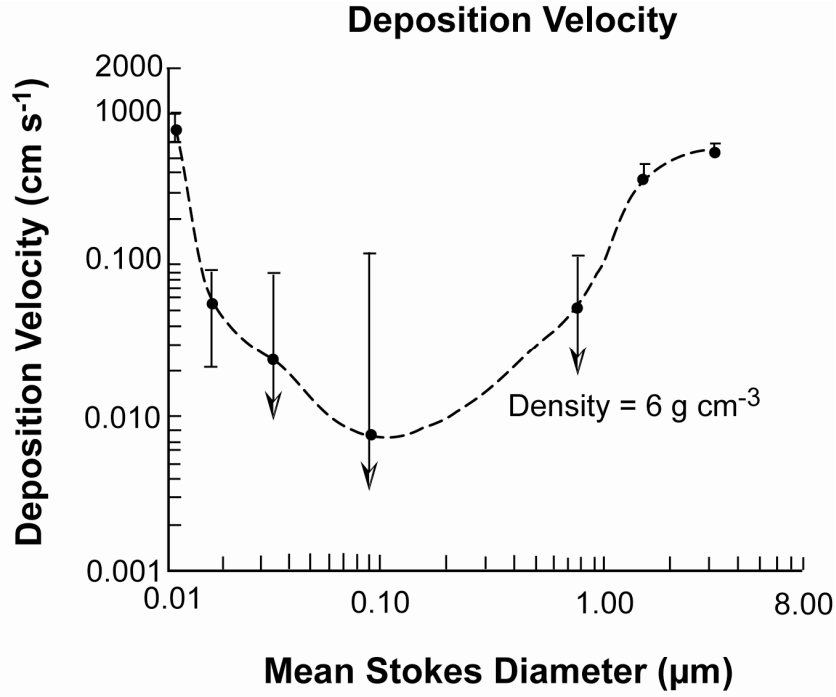

Figure 2-2. The deposition velocity plotted against the geometric mean Stokes diameter for particles with a density of 6 g/cm^{-3} (i.e., lead). Error bars are shown and the arrow indicates a negative value for the lower limit of uncertainty.

Source: Reprinted from Main and Friedlander (1990).

The size of particles can influence rates of wet deposition. Large particles are scavenged more efficiently. Lead, which is primarily in the submicron size range, does not undergo wet deposition as easily as many of the crustal elements (Davidson and Rabinowitz, 1992).

Conko et al. (2004) note a seasonal trend in wet deposition rates of lead. The highest concentrations were observed in the summer months, which the authors attribute to increased emissions from electric power plants. This contradicts the findings of Gélinas and Schmit (1998) who found the lowest deposition of lead occurred in the summer. The authors suspect that this is due to decreased traffic in the summer months.

Regional differences may affect deposition rates. Miller and Freidland (1994) observed wet deposition fluxes at an elevation of 1200 m that were almost twice as high as fluxes observed at low altitudes. This effect is observed because wet deposition of lead occurs almost exclusively through rainfall at low elevations, but cloudwater interception is an important factor

at high elevations in addition to rainfall. Comparing values for urban and rural sites, similar rates of wet deposition were observed indicating that lead is widely disbursed and is emitted by area sources (Conko et al., 1994).

Precipitation activity has been linked to variability in wet deposition rates. Intense rain showers had lower lead concentrations than slow, even rainfalls (Chow, 1978). Thunderstorms typically did not have detectable quantities of lead but occasionally produced very high levels.

The concentration of lead in rainfall does not appear to be correlated to the amount of time between rainfalls but meteorological conditions such as a thermal inversion preceding a rainfall may affect the lead content (Chow, 1978).

Lead in rainwater includes both dissolved and particulate material. Approximately 83% of lead in wet deposition samples was in a soluble form, compared to less than 10% in dry deposition samples (Gatz and Chu, 1986).

Typical concentrations of lead in precipitation are listed in Table 2-20. The table shows a pronounced downward trend with time presumably following the phase out of leaded fuel.

Table 2-20. Concentrations of Lead in Rainwater in the United States

Dates of Testing	Precipitation concentration (µg/L)	Cloudwater concentration (µg/L)	Source	
1966-1967	32.7	98.1	Miller & Friedland (1994)	(taken from Lazrus et al., 1970)
1971-1972	31.2	93.6	Miller & Friedland (1994)	(taken from Schlesinger & Reiners, 1974)
1975-1976	25.2	75.6	Miller & Friedland (1994)	(taken from Smith and Siccama, 1981)
1977-1978	15.6	46.8	Miller & Friedland (1994)	(taken from Smith and Siccama, 1981)
1982	17.0	51	Miller & Friedland (1994)	(taken from Scherbatskoy and Bliss, 1984)
pre-1982	19.7	n.a.	Davidson & Rabinowitz (1992)	(taken from Galloway et al., 1982)
1988-1989	1.9	5.4	Miller & Friedland (1994)	(taken from Miller and Friedland, 1991)
1998	0.47	n.a.	Conko et al. (2004)	

Source: Miller and Friedland (1994), Davidson and Rabinowitz (1992), and Conko et al. (2004).

Bulk Deposition

Bulk deposition is the rate of dry and wet deposition combined. It is typically sampled in open buckets or other open containers. This is often used to estimate the overall rate of atmospheric input to soil, surface water, or other terrestrial media. However, it is understood that dry deposition onto surrogate surfaces may differ greatly from dry deposition onto natural surfaces. The ratio of dry to wet deposition is 1.5, 0.4, and 0.25 in marine, rural, and urban areas respectively (Davidson and Rabinowitz, 1992). The ratio of dry deposition to wet deposition ranged between 0.1 and 0.5 in arctic regions (Davidson and Rabinowitz, 1992). In a literature survey, Hicks (1986) found that this ratio varied between 0.4 and 1.8.

2.3.3 Resuspension of Lead-Containing Soil and Dust Particles

The resuspension of soil-bound lead particles and contaminated road dust is a significant source of airborne lead. Here we focus on resuspension by wind and vehicular traffic although resuspension through other mechanical processes such as pedestrian traffic, agricultural operations, construction, and even raindrop impaction is possible. In general, mechanical stresses are more effective at resuspending particles than wind (Sehmel, 1980; Nicholson, 1988b). We begin with a discussion of resuspension by natural winds, then present information on the effect of vehicular traffic on resuspension.

Understanding the physics of resuspension requires analyzing the wind stresses on individual particles including frictional drag, form drag, gravitation, and the Bernoulli effect (Sehmel, 1980). Although this analysis can be accurate on a small scale, predicting resuspension on a large scale generally focuses on empirical data for continual soil movement due to three processes: saltation, surface creep, and suspension (Sehmel, 1980; Nicholson, 1988b). Saltation is the process by which particles in the 100-500 μm size range bounce or jump close to the surface. The low angle at which these particles strike the surface can transfer momentum to smaller particles allowing them to be suspended into the atmosphere (Sehmel, 1980; Nicholson, 1988b). Depending on soil conditions, saltation can be responsible for moving 50-75% of surface particles. Surface creep is the rolling or sliding motion of particles, which is induced by wind stress or momentum exchanged from other moving particles. This generally applies to large particles 500-1000 μm in diameter and moves 5-25% of soil by weight (Sehmel, 1980; Nicholson, 1988b). Suspension is the process that actually ejects particles into the air. This

affects particles smaller than 100 μm in diameter and moves 3-40% of soil by weight (Sehmel, 1980; Nicholson, 1988b).

Resuspension is often defined in terms of a resuspension factor, K, with units of m^{-1}, or a resuspension rate (Λ), with units of sec^{-1}. The resuspension factor was used in early research on reentrainment and is defined by:

$$K = \frac{C_{air}(\mu g / m^3)}{C_{soil}(\mu g / m^2)} \quad (2\text{-}7)$$

where C_{air} is the airborne concentration of a chemical species and C_{soil} is the surface soil concentration of the same species. K has significant limitations in that it is dependent both on the height at which C_{air} is measured and the depth to which C_{soil} is measured. This factor also assumes that all airborne material is a direct result of resuspended soil-bound material, which is not the case in most situations (Sehmel, 1980; Nicholson, 1988b). Additionally, K cannot be used if soil concentrations are not uniform across the area of interest (Nicholson, 1988b).

The resuspension rate, Λ, is the fraction of a surface contaminant that is released per time and is defined by:

$$\Lambda = \frac{R(\mu g / m^2 s)}{C_{soil}(\mu g / m^2)} \quad (2\text{-}8)$$

where R is the upward resuspension flux, and Λ has units of s^{-1}. Although Λ is also dependent on the depth to which soil concentrations are measured, the resuspension rate has a number of advantages over K. Most notably, it can be applied to non-uniform areas of soil contamination, and it allows for other sources of airborne contaminants. It cannot be determined experimentally and is usually deduced by fitting results to a numerical model of airborne dispersion and deposition for the pollutant of interest (Nicholson, 1988b). Resuspension rates are dependent on many factors including wind speed, soil moisture, particle sizes, the presence of saltating particles, and the presence of vegetation. Typical values for Λ can cover 9 orders of magnitude in the range of 10^{-12}-10^{-4} s^{-1} (Sehmel, 1980; Nicholson, 1988b).

Nicholson (1993) notes that Λ increases with increasing particle diameter because larger particles protrude farter into the turbulent air stream and the drag force increases more quickly

than the adhesive force. Furthermore, in a laboratory resuspension chamber, the yields of resuspended matter decreased approximately linearly with increases in the geometric mean particle sizes of the bulk soil (Young et al., 2002). Lead is associated with the smaller size ranges in the distribution of soil particles. Young et al. (2002) suggest that this is because the higher specific surface area of small particles means that there are higher contents of organic matter or Fe/Al oxides that serve as lead binding sites.

Saltation is a particularly important factor in determining resuspension rates. Saltation moves large quantities of soil particles and is highly efficient at ejecting particles into the airstream. Saltating particles rotate between 200 and 1000 revolutions/second and are ejected almost vertically (Sehmel, 1980). Saltating particles strike the surface at very small angles – almost horizontally – and cause an avalanching effect. In the absence of saltation, very little resuspension would occur at all (Sehmel, 1980; Nicholson, 1993). Because resuspension is driven by saltation and not the direct pick-up by wind, the size distribution of resuspended particles does not change with windspeed (Young et al., 2002).

Vehicular resuspension is the result either of shearing stress of the tires or turbulence generated by the passing vehicle (Nicholson, 1988b; Nicholson et al., 1989). This process can be particularly important since the most contaminated roadways tend to have the most traffic. As with wind resuspension, a number of factors can affect the rate of resuspension from vehicular motion. These factors include vehicle size, vehicle speed, moisture, and particle size.

Lead in street dust appears to have a bimodal distribution. The fine mode is likely from vapor phase condensation from combustion engines, while the coarse mode is from either vehicle wear or significant coagulation of smaller particles. Al-Chalabi and Hawker (1997) observed that in roadways with significant resuspension, lead concentrations were lower indicating either dispersion from the source or the scavenging of smaller lead particles by coarser particles. Abu-Allaban et al. (2003) similarly observed that lead in road dust tended to be in the coarse mode. Measurements performed in tunnel tests indicated that less than 17% of PM_{10} lead was smaller than 2.5 μm (Lough et al., 2005).

Resuspension may occur as a series of events. Short episodes of high windspeeds, dry conditions, and other factors conducive to resuspension may dominate annual averages of upward flux (Nicholson, 1988b, 1993).

The concentrations of lead in suspended soil and dust vary significantly. In suspended soils sampled near industrial emitters of lead, PM_{10}-bound lead varied between 0.012 and 1.2 mg Pb/ kg of bulk soil (Young et al., 2002). Tsai and Wu (1995) measured lead in airborne particles that was 30 times higher than lead in road dust. This enrichment factor was much higher than for other pollutants, which may indicate that lead is more easily resuspended than other contaminants. The fractions of lead in suspended dusts and soils are listed in Table 2-21.

Table 2-21. The Percentage of Lead in Resuspended Particulate Matter

Source	Location	Pb fraction of PM_{10} mass (%)	Pb fraction of $PM_{2.5}$ mass (%)	Reference
Paved road dust	urban	0.0161 ± 0.0031		Chow et al. (2003)
Paved road dust	urban	0.3 ± 0.03	0.4	Chow et al. (1994)
Paved road dust	urban		1.E-02	Gillies et al. (1999)
Paved road dust	rural	0.0057 ± 0.0028		Chow et al. (2003)
Unpaved road dust	rural	0.0058 ± 0.0073		Chow et al. (2003)
Unpaved road dust	rural	0.01		Chow et al. (1994)
Unpaved road dust	residential	0.0203 ± 0.0133		Chow et al. (2003)
Unpaved road dust	staging area	0.0043 ± 0.0008		Chow et al. (2003)
Agricultural soil		0.0063 ± 0.0059		Chow et al. (2003)
Agricultural soil		0.0031 ± 0.0025		Chow et al. (2003)
Agricultural soil		0.0062 ± 0.0034		Chow et al. (2003)
Agricultural soil		0.0024 ± 0.0082		Chow et al. (2003)
Agricultural soil		0.003 ± 0.0025		Chow et al. (2003)
Agricultural soil		0.01		Chow et al. (1994)
Playa dust	rural		1.E-03	Gillies et al. (1999)
Sand & gravel storage		0.02		Chow et al. (1994)
Construction site	urban		1.E-03	Gillies et al. (1999)

Source: Chow et al. (1994, 2003) and Gillies et al. (1999).

The contribution of resuspended soil and dust to the airborne burden may be significant. A source apportionment study in Boston indicated that soil resuspension increased the airborne concentration of lead by as much as 0.022 μg/m^3 in the fine mode (Thurston and Spengler, 1985). Isotopic measurements in Yerevan, Armenia credited resuspension of contaminated soil with 75% of the atmospheric lead in 1998 (Kurkjian et al., 2002). Calculations based on road dust emissions and lead weight fractions indicate that resuspension was responsible for ~40% of overall lead emissions to the South Coast Air Basin of California in 1989 (Lankey et al., 1998). Resuspension estimates based on modeling efforts for the same area suggest that resuspension contributed ~90% of overall lead emissions in 2001 (Harris and Davidson, 2005). Figures 2-3 and 2-4 demonstrate how air and soil concentrations are affected by long-term resuspension.

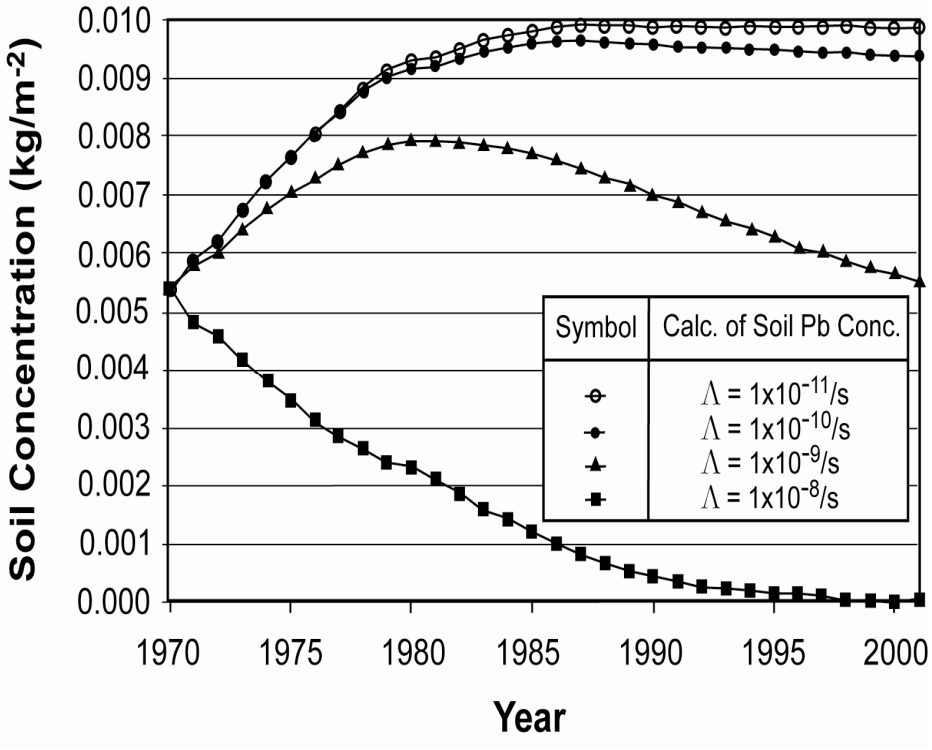

Figure 2-3. The modeled soil concentrations of lead in the South Coast Air Basin of California based on three resuspension rates.

Source: Reprinted from Harris and Davidson (2005).

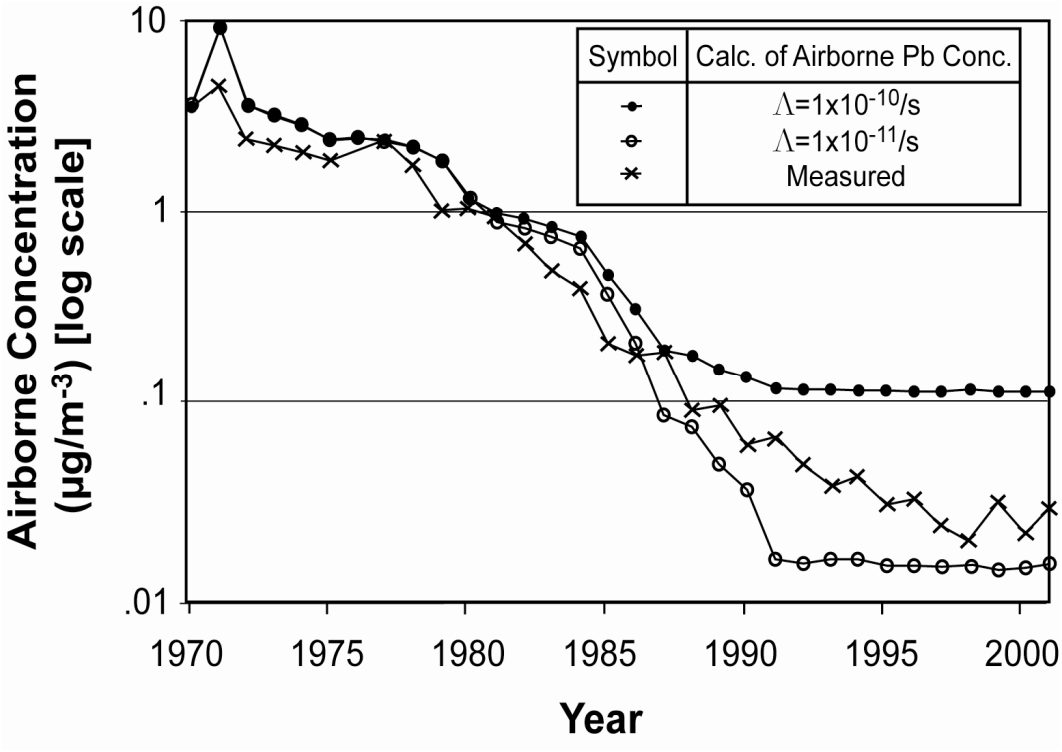

Figure 2-4. The modeled and measured airborne concentrations of lead in the South Coast Air Basin of California based on two resuspension rates.

Source: reprinted from Harris and Davidson (2005).

2.3.4 Runoff from Impervious Surfaces

The runoff of water from impervious surfaces may be a significant transport route for lead from urban areas to soil, waterways, and catchment basins. As water runs off roadways and buildings, it can become laden with dissolved and suspended matter. Dust on roadways contains a significant fraction of lead due to vehicle wear, vehicle emissions, road wear, fluid leakage, and atmospheric deposition. Lead in road dust is discussed in further detail in Sections 2.3.3 and 3.2 of this document. Additionally, lead-containing paints, gutters, roofing materials, and other housing materials may leach with rainfall.

Urban catchments in Fresno, California had highly elevated soil lead concentrations, which probably indicates high concentrations of lead in runoff waters (Nightengale, 1987). Basins in use since 1962, 1965, and 1969 had surface soil concentrations of 570, 670, and 1400 ppm, respectively. Nearby control soils had surface concentrations between 8.3 and 107.0 ppm.

Urban runoff released into a stream in State College, Pennsylvania caused significant spikes in lead concentrations (Lieb and Carline, 2000). Concentrations upstream of the release point were 1.50 µg/L. Downstream concentrations were 1.80 µg/L when there was no precipitation and averaged 14.60 µg/L during storm events.

The amount of lead that is removed from roadways and buildings by rainwater depends somewhat on the intensity of the storm. Experiments performed by Davis and Burns (1999) indicated that high intensity storms washed away significantly more exterior house paint than low-intensity storms. A separate set of experiments showed that the amount of lead contained in roadway runoff increased significantly with the length of the dry period prior to a rain event (Hewitt and Rashed, 1992).

Lead in runoff water is primarily in the particulate form with a very small fraction in the dissolved form (Hewitt and Rashed, 1992; Davis and Burns, 1999; Roger et al., 1998). Between 69% and 93% of lead washed from painted structures was in particulate form (Davis and Burns, 1999). More than 90% of lead in highway runoff from a rural highway in the UK was in the particulate phase (Hewitt and Rashed, 1992). Roger et al. (1998) observed that lead particles in a motorway catchment in France were typically smaller than 50 µm in diameter. Samples taken from road water samples also in France showed that most lead was in an inorganic, non-bioavailable form (Flores-Rodríguez et al., 1994).

The amount of lead from roadways varies by region, the rainfall intensity, maximum inflow, rainfall duration, and the antecedent dry weather period (Shinya et al., 2000). Measurements taken near a roadway in France showed that in runoff water, concentrations ranged between 0.46 and 4.57 g Pb/kg of suspended particulate matter (Roger et al., 1998). Another study of French roadways had an average lead content of 2.36 g Pb/kg of dried material (Flores-Rodríguez et al., 1994). Thirteen storm events studied at a heavily trafficked, rural highway in England showed mean lead contents of 181 µg/L (Hewitt and Rashed, 1992). Of this total, 16.2 ± 6.9 µg/L was in the dissolved phase and 165 ± 101 µg/L was in the particulate phase. An additional 0.36 µg/L was in an organic form. The mean concentrations of lead during four rain events studied near a roadway in Japan ranged between 17 and 39 µg/L (Shinya et al., 2000). The initial concentrations were higher, ranging from 130 to 567 µg/L. This indicates the presence of a first flush effect in which much of the contamination is removed within the initial period of rainfall. Hewitt and Rashed (1992) observed a similar downward trend in the lead

concentration with time. However, no first flush phenomenon was observed in the study performed by Taebi and Droste (2004), who looked at combined urban runoff transported to a mixed residential and commercial urban catchment in Iran. The concentrations of lead for each of 10 major rainfall events ranged between 0.018 and 0.558 µg/L. The arithmetic mean for all 10 events was 0.278 µg/L.

Studies of runoff from building materials showed high lead concentrations from painted wood and painted brick, particularly if the paint is more than 10 years old (Davis and Burns, 1999; Davis et al., 2001). The maximum concentrations of lead were 1900 µg/L and 28000 µg/L from wood and brick with painted exterior surfaces, respectively (Davis and Burns, 1999). Lead from paint is released into waters in both particulate and dissolved form. The concentrations in runoff from building surfaces are listed in Table 2-22.

Table 2-22. The Concentrations of Lead in Runoff From Building Surfaces

Substance	Geometric Mean (µg/L)	Median (µg/L)	Mean µg/L)	Range (µg/L)	Reference
Block (painted)	9.2	8.0	38	<2-590	Davis and Burns (1999)
Brick (painted)	22	16	580	<2-28000	Davis and Burns (1999)
Wood (painted)	43	49	170	<2-1900	Davis and Burns (1999)
0-5 yr. old paint	8.0	8.1	27	<2-370	Davis and Burns (1999)
5-10 yr. old paint	18	14	120	<2-2600	Davis and Burns (1999)
>10 yr. old paint	81	88	810	<2-28000	Davis and Burns (1999)
Roofs	6.0	5.2	38	<2-590	Davis and Burns (1999)
Residential roofs		2	1.5		Davis et al. (2001)
Commercial roofs		12	62		Davis et al. (2001)
Institutional roofs		64	64		Davis et al. (2001)

Source: Davis and Burns (1999) and Davis et al. (2001).

Matthes et al. (2002) studied runoff from lead sheet to simulate lead in gutters, roofs, piping, siding, and sculptures. Typical concentrations in runoff ranged between 700 and 3700 mg/L. This was attributed to the solubility of cerrusite (lead carbonate) and hydrocerrusite (lead hydroxy carbonate), which form on the surface of air-exposed lead.

The amount of lead removed by runoff events varies. Hewitt and Rashed (1992) estimate that approximately 8% of lead and 5% of organic lead emitted from vehicles is removed by highway drainage waters. Shinya et al. (2000) estimate that total lead loads for a roadway in Japan prior to four storm events ranged between 0.053 and 0.771 mg Pb/m^2. These storm events removed half of the load in 0.07 to 3.18 hours after the start of the rainfall event.

Davis et al. (2001) estimate the total annual loading of lead from all sources to be between 0.069 and 0.18 kg Pb/ha. They estimate that 80-90% of this is derived from runoff from buildings.

Lead corrosion (cerrusite and hydrocerrusite) dissolution rates from lead sheets were measured at 14.3 – 19.6 millimoles of lead/m^2 per year (Matthes et al., 2002).

2.3.5 Leaching of Soil Lead

Soil lead has some capacity to leach through the soil column, potentially contaminating ground water. Lead sorbs strongly to constituents of the soil matrix and is only weakly soluble in pore water, so the leaching of lead is a much slower process than the leaching of many other contaminants (Marcos et al., 2002; Zhang and Xu, 2003; Ünlü, 1998; Pang et al., 2002). The sorbing capacity of the soil and the solubility of the contaminants can be affected by the hydraulic conductivity of the soil, the composition of the soil solution, the content of the soil organic matter, the content of the soil clay minerals, soil pH, microbial activity, preferential flow through plant root channels and animal holes, and geochemical reactions (Rhue et al., 1992; Elzahabi and Yong, 2001). The experiments of Erel et al. (1997) on soil columns indicate that anthropogenic lead is more readily available for leaching than lead that naturally occurs in the soil.

Lead can bind to many different surfaces in the heterogeneous soil matrix. This adsorption greatly affects mobility and is dependent on the characteristics of the soil and lead compounds. Lead is partitioned between the soil water solution, precipitated forms, secondary Fe or Mn oxides, carbonates, organic matter, sulfides, or the surfaces of clay, humus, or silicate

particles (Badawy et al., 2002; Venditti et al., 2000; Cajuste et al., 2000; Erel and Patterson, 1994). The most labile fraction of lead is adsorbed to the surfaces of colloid soil particles, which may include organic matter, clay, oxides, or carbonates (Erel et al., 1997). Lead that was leached from a limestone soil during a sequential fractionation procedure was exclusively in the iron/manganese oxide form (Hee, 1994). A study of industrially contaminated soils found that between approximately 50% and 60% of the lead was not susceptible to leaching during any phase of a sequential fractionation procedure (Cajuste et al., 2000). The remaining lead was found primarily in the carbonate and Fe-Mn oxide fractions with sizeable amounts in the organic and exchangeable phases. None of the lead was water soluble. Maskall and Thornton (1998) also observed a high fraction of lead in the carbonate form in highly contaminated soil. The unusual presence of carbonate-bound lead is probably due to the formation of cerrusite ($PbCO_3$) in soils contaminated with calcareous slag wastes (Maskall and Thornton, 1998). Lead migration in this contaminated soil was associated with Fe-Mn oxides. A third contaminated site was tested by Jing et al. (2004). These soils showed 57% of lead in the Fe-Mn oxide form, 29% in the carbonate form, and just 5% in the residual, soil-bound form.

A high chlorine content in the soil has been shown to increase lead leaching (Ünlü, 1998). Chloride complexation with lead enhances lead solubility.

The pore-water velocity is inversely proportional to sorption rates. At low flow, the longer retention times lead to more complete sorption of lead to soil particles (Pang et al., 2002).

In laboratory experiments on soil columns, transport of lead was enhanced by the introduction of soil colloid suspensions (Karathanasis, 2000). Colloids increased transport of not only colloid-bound lead but also dissolved lead. Colloid transport was enhanced by increasing the colloid surface charge, increasing the pH, increasing the amount of organic carbon, increasing the soil macroposity, decreasing the colloid size, and decreasing the Al, Fe, and quartz contents (Karathanasis, 2000). Colloid binding and co-transport of lead are important mechanisms for lead migration but colloids also enhance the flow of lead through physical blockage from exchange sites, competitive sorption, and organic complexation (Karathanasis, 2000). Denaix et al. (2001) observed that most of the lead-transporting colloids in an acidic, loamy soil were biological in nature. The lead concentration in the colloid fraction was not correlated with pH, colloidal organic carbon contents, or colloidal silicon concentrations (Denaix

et al., 2001). Approximately 50% of the total lead transfer in these experiments was attributed to colloidal transfer.

At low pH, metal species bound to carbonates, hydroxides, and other soil matrix components are more likely to dissolve into solution (Maskall and Thornton, 1998; Elzahabi and Yong, 2001; Badawy et al., 2002). This increases the rate of lead migration through the soil. The experiments of Jing et al. (2004), which follow eight different leaching protocols, suggest that pH is the primary factor in determining the concentration of lead in leached solution. At pH >12, lead forms soluble hydroxide anion complexes and leaches out of the soil column. At pH between 6 and 12, lead leachibility is low due to adsorption and precipitation. At pH <6 free Pb ions leach into the pore water and are removed from the soil columns. Rhue et al. (1992) observed that organic lead species Me_2Pb^{2+} and Et_2Pb^{2+} were absorbed best at pHs of 6.2 and 7.2, respectively. Sorption decreased at pH <5 and >8.2 (Rhue et al., 1992).

A partition coefficient, K_d, is often used to describe the susceptibility of lead to leaching. This value is used to compare the fractionation of a contaminant between liquid and solid forms. K_d is defined by the following equation:

$$K_d = S/C' \qquad (2\text{-}9)$$

where S is the total concentration of lead adsorbed in the solid phase, and C' is the concentration of lead in pore water solution (Elzahabi and Yong, 2001). K_d increases with increasing pH (up to 7.0) and increasing distance from the leachate source (Elzahabi and Yong, 2001; Sheppard and Sheppard, 1991). K_d decreases with an increase in the influent heavy metal concentration and the degree of saturation (Elzahabi and Yong, 2001). The highest value of K_d appears to be near the source of lead contamination. Values of K_d in the literature cover many orders of magnitude between 1.20 L/kg and "infinity" (when no lead can be detected in pore water). These values are listed in Table 2-23.

The rate of migration through the soil has been estimated in many different studies. Using lead isotopes, Erel et al. (1997) estimate the rate of lead migration to be 0.5 cm/year in soils collected from rural locations in Israel. Sheppard and Sheppard (1991) measured the rate of flow through spiked soils, which were highly acidic and had a low organic matter content. These soils, which were especially susceptible to leaching, exhibited migration rates of 0.3 cm/day

Table 2-23. Soil/Water Partition Coefficients for Several Different Soils and Conditions

Kd (L/kg)	pH	Beginning Soil Water Content (%)	Soil Type	Reference
12.68-∞	4.0	26.69	illitic (spiked)	Elzahabi and Yong (2001)
3.23-∞	4.0	28.20	illitic (spiked)	Elzahabi and Yong (2001)
1.20-∞	3.5	26.29	illitic (spiked)	Elzahabi and Yong (2001)
1.36-∞	3.5	26.32	illitic (spiked)	Elzahabi and Yong (2001)
~6000	n.a.	n.a.	brown pseudopodzolic	Alumaa et al. (2002)
~3000	n.a.	n.a.	rendzina	Alumaa et al. (2002)
~5000	n.a.	n.a.	gley podzolic	Alumaa et al. (2002)
20	4.9	n.a.	acidic (low-organic-matter sand)	Sheppard and Sheppard (1991)
9000	4.8	n.a.	sphagnum peat	Sheppard and Sheppard (1991)
92.99	4.45	n.a.	mining site	Merrington and Alloway (1994)
14.25	4.45	n.a.	mining site	Merrington and Alloway (1994)
125.58	5.01	n.a.	mining site	Merrington and Alloway (1994)
95.51	5.01	n.a.	mining site	Merrington and Alloway (1994)
1330±200	3.0-4.0	n.a.	acidic (high-organic-matter peat)	Deiss et al. (2004)

Source: Elzahabi and Yong (2001), Alumaa et al. (2002), Sheppard and Sheppard (1991), Merrington and Alloway (1994), and Deiss et al. (2004).

during the first year of experiments. The migration rate appeared to slow down in subsequent years. Cores taken at smelting sites used during the Roman era, medieval times, and the 18th century underwent sequential extraction (Maskall and Thornton, 1998). The estimated lead migration rates at the Roman, medieval, and 18th century sites were 0.07–0.54 cm/year, 0.31-1.44 cm/year, and 0.11–1.48 cm/year, respectively.

Mass balance calculations of Miller and Friedland (1994) suggest migration rates of 0.11 cm/year and 0.29 cm/year through the organic horizons of spruce-fir and northern hardwood forests, respectively. Similar calculations by Kaste et al. (2003) at the same site predicted that anthropogenic lead will take ~60 and ~150 years to be transported through the organic horizon in

the deciduous and spruce-fir forests, respectively. The difference in response times for the two forests may be due to differences in the litter depth and/or in the rate of litter decomposition. Soil tested from a car battery salvage facility showed a significantly greater lead concentration in the leached solution than in a reference soil (Jensen et al., 2000). Concentrations in the leached solution went as high as 8000 µg/L. Other industrially contaminated soils did not show such high rates of leaching, but these other soils had nearly neutral pHs.

Isotopic ratios in soil cores in the Sierra Nevada, California showed that 21% of lead at a depth of 30 cm had anthropogenic origins and had migrated from the surface (Erel and Patterson, 1994). The remaining 79% of lead at this depth was naturally occurring.

Physical mixing of soils through animal activity may also increase the rate of lead migration. Mace et al. (1997) observed a significant decrease in lead transport time through soil as a result of rodent activity in a southern California location.

Vilomet et al. (2003) used isotopes to trace the leaching of lead from a landfill into groundwater in France. The active landfill has been in use since 1900 and has no bottom liner. Detectable quantities of leached lead were observed as far as 4600 m downgradient (Vilomet et al., 2003).

2.3.6 Transport in Aquatic Systems

Chemical, biological, and mechanical processes govern the cycling of lead in aquatic environments. Here we focus on the exchange between sediment and surface water, which is affected by many different factors including salinity, the formation of organic complexes, redox conditions, and pH (Arakel and Hongjun, 1992).

Lead enters surface waters from a number of sources. Atmospheric deposition is the largest source, but urban runoff and industrial discharge are also significant (Peltier et al., 2003; Hagner, 2002; Perkins et al., 2000). As expected, concentrations in surface waters are highest near sources of pollution.

The dispersal of lead in waterways is relatively quick. If lead is emitted into waterways as a point source, water concentrations decrease rapidly downstream of the source (Rhoads and Cahill, 1999; Hagner, 2002; Kurkjian et al., 2004; Peltier et al., 2003). Lead is removed from the water column through flushing, evaporation, or sedimentation (Schell and Barnes, 1986).

Kurkjian et al. (2004) note that first order approximations of concentrations of non-conservative pollutants (such as lead) can be made by using the exponential decay curve:

$$C = C_o e^{-kx} \quad (2\text{-}10)$$

where C is the pollutant concentration, C_o is the concentration at the source, x is the downstream distance from the source, and k is the decay rate in km^{-1}. For the Debed River in Armenia, Kurkjian et al. (2004) found that a decay rate of 0.57 km^{-1} provided the best fit to measured lead concentrations.

Metals in waterways are transported primarily as soluble chelates and ions, constituents of particulate matter, or by adsorption onto suspended organic or inorganic colloids (Arakel and Hongjun, 1992). The last two are the most important for lead. The predominant chemical forms of lead that interact with aqueous ecosystems are PbO and $PbCO_3$ (Schell and Barnes, 1986). Lead is adsorbed on colloids that are typically secondary clay minerals, Fe-Mn oxides or hydroxides, or organic compounds (Arakel and Hongjun, 1992). The concentration of lead appears to increase with increasing salinity (Arakel and Hongjun, 1992).

Schell and Barnes (1986) describe water columns as "transient reservoirs" for pollutants. They found mean residence times for lead in two lakes and a reservoir to be between 77 and 250 days, although it should be noted that residence times tend to be shorter in turbulent waterways. Lead concentrations in water are attenuated by the presence of $Al(OH)_3$ precipitation, which is responsible for an estimated 54% of total lead loss, and by the adsorption of lead onto other particles which settle out of the water column, which makes up the other 46% of lead loss (Kurkjian et al., 2004). Schell and Barnes (1986) measured sedimentation rates for anthropogenic lead, which ranged between 0.0360 g cm^{-2} a^{-1} and 0.0644 g cm^{-2} a^{-1}.

The concentration of lead in sediment roughly follows the concentration of lead in overlying water (Kurkjian et al., 2004; Rhoads and Cahill, 1999). Thus lead concentrations in sediment are highest near sources and decrease downstream.

Lead preferentially sorbs onto small particles rather than large particles. Small grain sizes and the larger surface area per unit weight lead to greater potential for adsorption (Rhoads and Cahill, 1999). Concentrations of metals increase approximately logarithmically with decreasing particle size.

Organic matter in sediment has a high capacity to accumulate trace elements. High humic levels may lead to greater lead contamination in sediments (Rhoads and Cahill, 1999; Kiratli and Ergin, 1996).

Sulfides are another potential source of lead adsorption. This is especially true under anoxic conditions (Kiratli and Ergin, 1996; Perkins et al., 2000). An increase in the amount of sulfide in pore water was shown to decrease the dissolved concentration of lead (Peltier et al., 2003).

Lead in sediment can also be sequestered on iron or manganese oxides (Peltier et al., 2003; Gallon et al., 2004; Schintu et al., 1991). These forms may make lead susceptible to recycling into the overlying water column (Schintu et al., 1991).

Lead appears to be relatively stable in sediment. It has a very long residence time, and many studies suggest that lead is not mobile in the sediment. However, many other studies suggest that lead-containing particles can be remobilized into the water column (Ritson et al., 1999; Steding et al., 2000; Hlavay et al., 2001; Kurkjian et al., 2004; Peltier et al., 2003; Gallon et al., 2004). For example, Steding et al. (2000) observe that isotopic concentrations of lead in the San Francisco Bay match those of leaded gasoline from the 1960s and 1970s, which may indicate that recontamination by sediment may be a significant source of lead to overlying waters. Ritson et al. (1999) similarly observed that there was a negligible reduction in lead concentrations in the San Francisco Bay despite the closing of a nearby lead smelter, the implementation of municipal effluent controls, and the elimination of lead additives to gasoline. That concentrations have remained high may suggest recycling of sediment lead. Similarly, in a study of water concentrations in the North Sea, concentrations of lead did not decrease significantly with the elimination of major sources (Hagner, 2002). This also may indicate continued high rates of atmospheric deposition or cycling of lead stored temporarily in sediment.

Modeling efforts of Gallon et al. (2004) indicate that processes that resuspend sediment such as diffusion, bioturbation, and bioirrigation are small compared to sedimentation of colloidal particles. Kurkjian et al. (2004) suggest a correction factor for equation (9) to account for the contribution of lead from sediment.

$$C = C_o e^{-kx} + I_s \qquad (2\text{-}11)$$

where I_s is the amount of lead that is resuspended into the water column. Depending on the region of the river under discussion, the authors extrapolated I_s values in the range of 1.3-2.8 μg Pb/L.

2.3.7 Plant Uptake

Plants that take up lead can be a source of lead exposure for wildlife, livestock, and humans that consume contaminated plants. A more thorough review of soil lead extraction by plants and subsequent effects on ecosystem health will be addressed in Chapter 8.

Plants grown in soils contaminated by mine spoils (e.g., Cobb et al., 2000), smelting operations (e.g., Barcan et al., 1998), sludge amendments (e.g., Dudka and Miller, 1999), contaminated irrigation water (e.g., Al-Subu et al., 2003), or lead-containing agrochemicals (e.g., Azimi et al., 2004) will have higher than natural concentrations of lead. In general, higher concentrations of lead in soils resulting increased lead levels in plants.

Although the transfer of soil lead to plants is generally small, all plants accumulate soil lead to some degree (Finster et al., 2004). The rate of uptake is affected by plant species, soil conditions, and lead species.

Of all the factors affecting uptake, pH is believed to have the strongest effect (Dudka and Miller, 1999). Acidic soils are more likely to have lead in solution and therefore available for absorption. This is sometimes attenuated by liming.

Most lead in plants is stored in roots and very little is stored in fruits (e.g., Finster et al., 2004; Cobb et al., 2000). Of 33 edible plants grown in urban gardens, roots had a median concentration that was 12% of the soil lead concentration (Finster et al., 2004). Shoot lead, when it was detectable, was just 27% of root lead.

Root vegetables seem the most prone to lead uptake followed by leafy vegetables (Dudka and Miller, 1999; Finster et al., 2004). Fruits and grains do not seem as susceptible to lead contamination.

Metals that are applied as salts (usually as sulfate, chloride, or nitrate salt) are accumulated more readily than the same quantity of metal added via sewage sludge, flue dust, or fly ash (Dudka and Miller, 1999). This is likely because metal salts lead to the formation of metal chloride complexes and ion pairs, which can increase metal diffusion and subsequent plant uptake.

2.3.8 Routes of Exposure for Livestock and Wildlife

There are many routes of exposure including food ingestion, drinking water, and inhalation for terrestrial organisms. For aquatic organisms, the main routes of exposure are food ingestion and water intake. Thus, it is often difficult to determine the original source of an organism's lead burden. A few representative studies are summarized here, which have analyzed routes of lead exposure for non-human animals. For a discussion of health effects, toxicity, and lead concentrations in animal tissue, the reader is directed toward Chapters 8 and 9 of this document.

Lead concentration of plants ingested by animals is primarily a result of atmospheric deposition of lead particles onto plant surfaces rather than uptake of soil lead through plant roots (Steinnes, 2001; Palacios et al., 2002; Dudka and Miller, 1999). The uptake of lead by the lowest trophic levels – invertebrates, phytoplankton, and krill for example – are some of the most important avenues for introducing lead into food chains (Pilgrim and Hughes, 1994; Sanchez-Hernandez, 2000; Hagner, 2002).

Some of the highest levels of lead exposure in animals occur near major sources like smelters. In two studies of horses living near smelters, the estimated ingestion rate was in the range of 2.4 to 99.5 mg Pb/kg body weight per day (Palacios et al., 2002) and 6.0 mg Pb/kg body weight per day (Liu, 2003). Both exposure rates were well above the estimated fatal dose for horses. Sheep grazing near smelters were similarly poisoned (Liu, 2003; Pilgrim and Hughes, 1994). Installation of pollution controls at a lead smelter in Slovenia greatly reduced the amount of lead in nearby vegetation and the blood lead levels of cows grazing on this vegetation (Zadnik, 2004). Lead concentrations in topsoil at this site did not decline in the 20 years since the pollution controls were implemented.

The amount of lead entering the food chain depends highly on the species of the animal, the species of their food, and where the organisms live. A study of sheep living in the southernmost part of Norway, which is the most polluted part of the country, showed a strong correlation between liver lead concentrations and moss concentrations (Steinnes, 2001). The sheep fed almost exclusively on a grass that picks up atmospherically deposited lead easily. Correspondingly high levels were also observed in hare and black grouse in this region. Similarly, a study of lead concentrations in raccoon tissues showed much higher concentrations in urban raccoons that rural raccoons (Khan et al., 1995). This may be because urban raccoons

are exposed to higher air concentrations, ingest human refuse, or frequently visit storm sewers. In general, ruminant animals appear to be more resistant to lead ingestion than monogastric animals (Humphreys, 1991).

Lead levels are somewhat elevated even in Antarctic animals (Sanchez-Hernandez, 2000). Antarctic food systems are supported by krill (*Euphausia superba*), which is the primary food source for organisms in higher trophic levels. Lead concentrations measured in *E. superba* were in the range of 0.17-12.0 ppm by dry weight. This is probably elevated above natural levels due anthropogenic input (Sanchez-Hernandez, 2000).

Acute lead poisoning observed in Laysan albatross (*Phoebastria immutabilis*) chicks was traced to the direct ingestion of paint chips by using isotopic analysis (Finkelstein et al., 2003). Blood lead levels in *P. immutabilis* at the Midway Island National Wildlife Refuge had a geometric mean of 190 µg/dL. *P. immutabilis* chicks at a reference site had blood lead levels of 4.5 µg/dL.

Contamination in mammals and fish livers was shown to be higher in highly polluted coastal zones than in the open sea (Hagner, 2002). In foraminifers, which are meiobenthic organisms, high sediment concentrations corresponded to high tissue concentrations. Sediment concentrations were 10-20 times higher than foraminifer concentrations. Fish take in lead either in their food or in water through their gills. The relative importance of these two mechanisms depends largely on the fish species. A literature survey suggests that there has been no observable decrease in fish muscle and liver concentrations in twenty years in marine or freshwater environments (Hagner, 2002). Lead concentrations in the harbor porpoise (*Phocoena phocoena*) appear to increase with the age of the animal. This was not true for the common seal (Phoca vitulina) (Hagner, 2002). Shrimp (*Palaemonetes varians*) were shown to absorb 4-8% of the lead content of its prey (Boisson et al., 2003). Between 52% and 57% of the lead accumulated from food was irreversibly retained in *P. varians* tissue. Just 2% of dissolved lead accumulated from water was retained in tissue (Boisson et al., 2003).

2.4 METHODS FOR MEASURING ENVIRONMENTAL LEAD

The previous 1986 AQCD (U.S. Environmental Protection Agency, 1986) contained a detailed review of sampling and analytical methods for lead in environmental media. Included in

that document were discussions of site selection criteria, sampling methods, sample preparation, and analysis techniques. Furthermore, the document included discussion of sampling of lead emissions from mobile and stationary sources. In this section, we present a brief summary of sampling and analysis of lead. For a more comprehensive discussion, the reader is referred to the 1986 Lead AQCD.

Emissions can be estimated from measurements at sources using grab samples, periodic samples, or continuous monitoring. Determining the rate of emissions requires knowing both the fluid flow rate and the concentration of lead in the fluid, usually air or water. Thus it is much easier to measure emissions from stacks than it is to measure fugitive, diffuse, or nonpoint emissions (Frey and Small, 2003).

Much of the recent improvement in measurement of lead emissions from sources is due to better sampling and analytical equipment. For example, better dilution tunnels can provide reliable samples from in-stack sampling, and improved analytical methods such as inductively couple plasma mass spectrometry permit determination of lead at lower levels than in years past. This means it is possible to obtain data from short sampling runs, permitting better time resolution.

Ambient air sampling for lead generally uses filter media, such as Teflon, or impactors that fractionate the airborne particles into different size ranges. Collection of samples of water, food, dust, or soil for lead determination can be performed in acid-washed containers. Wet deposition can be collected using precipitation buckets that seal tightly immediately before and after rain. Dry deposition on land can be sampled using surrogate surfaces such as Teflon plates (Davidson et al., 1985; Davidson and Wu, 1990), or alternatively by leaf-washing (Lindberg and Lovett, 1985) or sampling throughfall precipitation that washes previously deposited lead off the vegetation and onto the forest floor (Wu et al., 1992b). Dry deposition onto bodies of water is more difficult to estimate, usually requiring airborne concentrations used with deposition velocity estimates (Zufall and Davidson, 1997). Subsequent analysis of all of these samples can be performed by atomic absorption spectrometry, neutron activation analysis, x-ray fluorescence, or proton-induced x-ray emission (Koutrakis and Sioutas, 1996), or by inductively-coupled plasma mass spectrometry (ICP-MS) (U.S. Environmental Protection Agency, 1991).

Recently developed single-particle instruments can identify which particles contain lead, and what other elements are present in the same particle. Information on the size of the particle

is also provided (Pekney et al., 2006; Silva and Prather, 1997). Although such instruments are not able to determine the precise mass of lead in each particle, they can provide valuable data on the characteristics of particles that contain lead from individual sources or source categories. Such "fingerprinting" methods can be used to identify sources of lead-containing particles in the environment.

2.5 SUMMARY

For most of the past 50–60 years, the primary use of Pb was as additives for gasoline. Leaded gasoline use peaked in the 1970s, and worldwide consumption has declined since (Nriagu, 1990).

Currently, the major use of Pb in the United States is in lead-acid batteries, for which the demand is increasing (Socolow and Thomas, 1997). Other major uses are for glass, paints, pigments, and ammunition. United States consumption of Pb is shown in Figure 2-5. The consumption reached ~1.4 million metric tons per year in the mid 1990s (Socolow and Thomas, 1997). Approximately 910,000 metric tons of this was secondary production, indicating high rates of Pb recycling.

The largest source of Pb emissions was leaded gasoline throughout the 1970s and 1980s. The largest emitters are now in the manufacturing sector, which includes lead-acid battery plants, smelters, lead-alloy production facilities, and others (Harris and Davidson, 2005). These emissions are not confined to the air — approximately 90 facilities nationwide generate 90% of the lead-containing solid hazardous waste (Chadha et al., 1998). Natural sources of Pb are insignificant in comparison to anthropogenic sources. Nationwide air emissions in 2000 were estimated as 1885 metric tons from metals processing, 758 metric tons from incineration, 513 metric tons from transportation, primarily from avgas-fueled aircraft, 439 metric tons from fuel combustion for utility generation as well as industrial and commercial purposes, 198 metric tons from Pb oxide and pigment production, and 48 metric tons from other processes (U.S. Environmental Protection Agency, 2003).

Emission inventories for Pb have significant omissions and discrepancies (Harris et al., 2005; Chadha et al., 1998). Thus, the data above are probably a lower limit for Pb emissions. Research into constructing detailed and accurate databases of Pb emissions is needed.

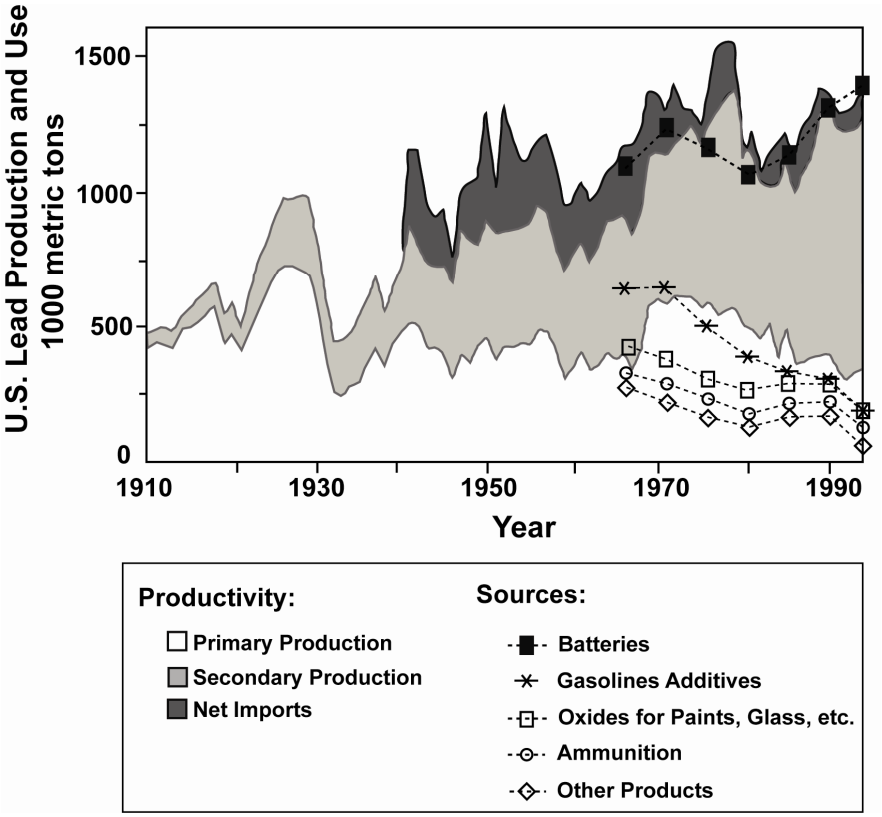

Figure 2-5. U.S. consumption of lead since 1910.

Source: Socolow and Thomas (1997).

The U.S. EPA Trends Report provides analysis of the available data on Pb emissions through the year 2002 (http://www.epa.gov/airtrends/lead2/html) (U.S. Environmental Protection Agency, 2003). Figure 2-6 shows the observed decline in estimated Pb emissions.

Measurements conducted in any ecosystem worldwide show some level of lead contamination. Anthropogenic Pb reaches these ecosystems through many possible transport routes, some of which are shown in Figure 2-7.

Air is the major transport route for lead emissions. Deposition of airborne pollutants to surfaces has been observed in the most remote places on Earth, including the Arctic and Antarctic. Mass balance calculations performed on an agricultural plot in France indicate that atmospheric deposition is the dominant source of lead to soil even when lead-containing

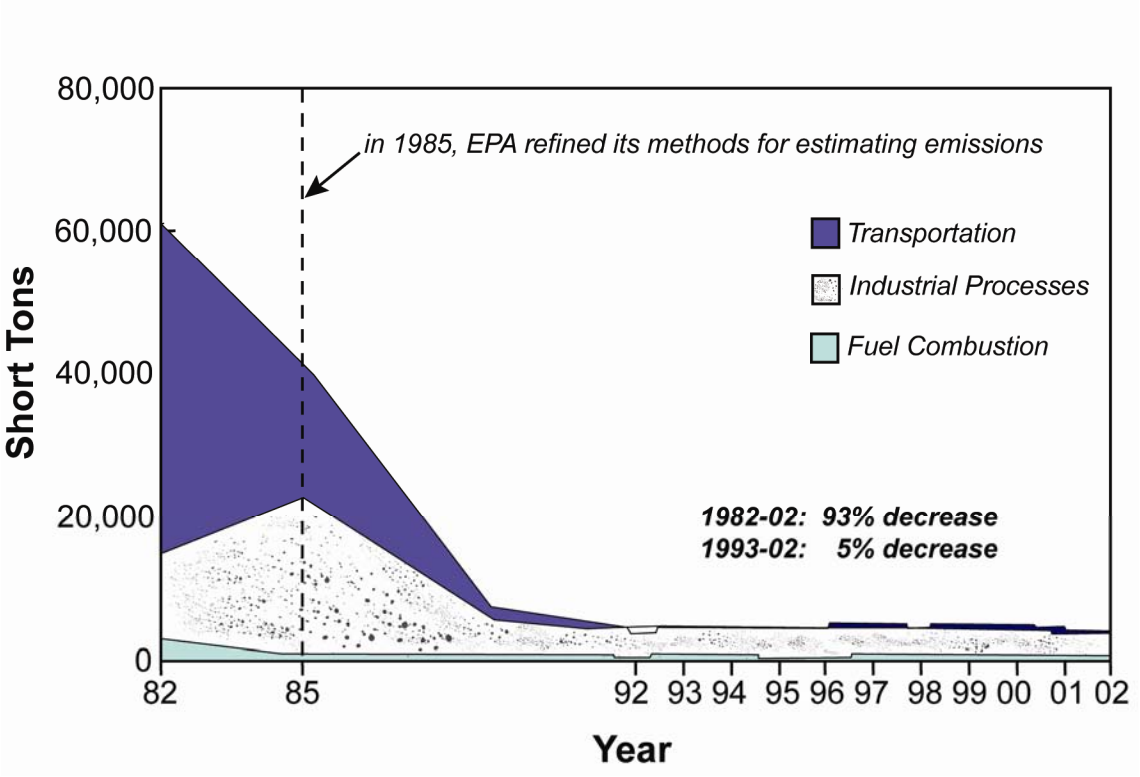

Figure 2-6. Trends in U.S. air lead emissions, 1982-2002.

Source: U.S. Environmental Protection Agency (2003).

fertilizer is applied (Azimi et al., 2004). However, on a local scale solid waste disposal or mine tailings may be the predominant source of soil lead.

A rigorous comparison of resuspension, leaching, and plant uptake "removal" rates for soil lead has not been undertaken. Resuspension of lead-containing particles is likely the dominant removal mechanism from surface soil when soil pH is high. Leaching may dominate when soil pH is low. Leaching of lead through soil occurs more rapidly than uptake to pea or wheat crops (Azimi et al., 2004). More research is needed to compare removal rates for other plants with soil lead migration and resuspension rates.

Surface waters are contaminated through several routes. On a global scale, sediment resuspension and wet and dry deposition are the predominant contributors to lead concentrations in surface water. On a local scale industrial effluent and urban runoff may dominate.

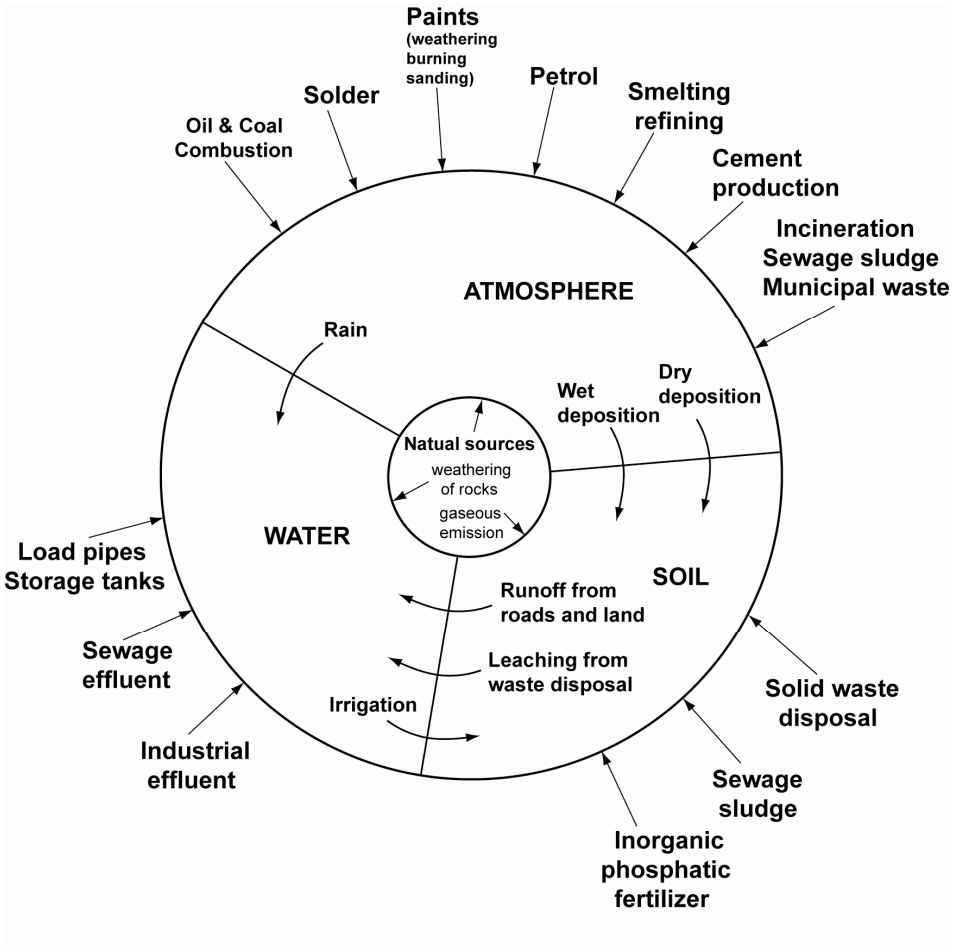

Figure 2-7. Transport pathways for lead in the environment.

Source: Reprinted from Zabel (1993).

The major routes of lead transport into the food chain appear to be ingestion of contaminated plants, ingestion of contaminated water, and inhalation of contaminated air. Research into the relative importance of each of these transport routes is needed.

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3. ROUTES OF HUMAN EXPOSURE TO LEAD AND OBSERVED ENVIRONMENTAL CONCENTRATIONS

3.1 EXPOSURE: AIR

3.1.1 Observed Concentrations – Indoor

Given the large amount of time people spend indoors, exposure to lead in dusts and indoor air can be significant. For children, dust ingested via hand-to-mouth activity is a more important source of lead exposure than inhalation (Adgate et al., 1998; Oliver et al., 1999). However, dust can be resuspended through household activities (e.g., Ferro et al., 2004), thereby posing an inhalation risk as well. The particle size of "dust" is not well defined, although 50 μm or 75 μm in diameter is sometimes given as an upper limit.

Lead in housedust is from a number of different sources. In general particulate matter can originate inside the home from sources such as smoking and cooking. However, lead appears to come from sources outside the home (Jones et al., 2000; Adgate et al., 1998). A chemical mass balance study in Jersey City, New Jersey observed that crustal sources contributed almost half of the lead in residences, lead-based paint contributed about a third, and deposition of airborne lead contributed the remainder (Adgate et al., 1998). Residential concentrations measured at the Bunker Hill Superfund Site in northern Idaho indicate that the concentration in houses depends primarily on the neighborhood soil concentration (von Lindern et al., 2003). However factors such as household hygiene, the number of adults living in the house, and the number of hours children spend playing outside were also shown to affect concentrations.

Living near a smelter or a mine contributes significantly to the lead load in residences (Rieuwerts and Farago, 1995; Rieuwerts et al., 1999). Homes of mine and smelter employees tend to have lead levels elevated above those of nearby houses indicating that lead can be transported into homes via workers (Rieuwerts et al., 1999).

Renovation and especially old paint removal can greatly increase lead levels inside the home (Mielke et al., 2001; Laxen et al., 1987; Jacobs, 1998). Removal of exterior paint via power sanding released an estimated 7.4 kg of lead as dust, causing lead levels inside one house to be well above safe levels (Mielke et al., 2001).

Lead concentrations are likely elevated somewhat in houses of smokers. In a nationwide (U.S.) study, blood lead levels were 38% higher in children who exhibited high cotinine levels, which indicate high second hand smoke exposure (Mannino et al., 2003). Lead is present both in tobacco and tobacco smoke, although lead concentrations in tobacco have fallen with decreases in the airborne concentration (Mannino et al., 2003). In a study performed in the UK, lead in house dust tended to be bound to the carbonate or Fe-Mn oxides (Feng and Barratt, 1994).

Concentrations of lead in house dust, school dust, and nursing home dust are shown in Table 3-1.

Metal-cored candlewicks have posed an additional significant source of indoor lead. The U.S. Consumer Product Safety Commission banned the use of metal-cored candlewicks that contain more than 0.06% lead as of October 15, 2003 (USGS, 2003). However, prior to this time, emissions of lead from metal-core wicks were measured in the range of 0.5-66 μg/hour according to one study (Nriagu and Kim, 2000) and 100-1700 μg/hour according to another study (Wassan et al., 2002). In homes where such candles were burned, airborne concentrations could have been well above ambient levels.

An additional concern is attic dust, or dust found in roof cavities. Significant deposits of atmospheric lead can build up in these spaces. This dust can seep into living spaces through ceiling decorative artwork, cracks between the wall and ceiling, electric light fittings, wall vents, or exhaust, roof, and ceiling fans (Davis and Gulson, 2005). Additionally, renovations, housing additions, ceiling collapses, and storm damage can produce large plumes of attic dust (Davis and Gulson, 2005).

Studies comparing lead concentrations in attic dust with house age showed an excellent correlation between lead levels and ambient air concentrations throughout the lifetime of the house (Chiaradia et al., 1997; Ilacqua et al., 2003). Attic dust may even serve as a proxy for predicting historic ambient concentrations although the resolution on such calculations would be low. Attic dust concentrations measured in Australia were an order of magnitude higher in houses near a copper smelter (Chiaradia et al., 1997). However, isotopic analyses showed that alkyl-lead additives were the dominant source of lead contamination in attic dust overall. The geometric mean concentration of lead measured in attics in Sydney was 1660 ppm near industrial sites, 1173 ppm near semi-industrial sites, 447 ppm in non-industrial sites, and 16 ppm in background, crustal materials (Davis and Gulson, 2005).

Table 3-1. Concentrations of Lead in Indoor Dust

Concentration of Lead (ppm)	Location	Reference	
503 (mean)	Edinburgh Scotland	Laxen et al. (1987)	
308 (median)			
43-13,600			
9 (geometric mean)	Various parts of Denmark	Jensen (1992)	
1.5-48.9	UK	Feng and Barratt (1994)	
117-362			
1598	Helena and Silver Valleys, US (near 2 Pb smelters)	Schilling and Bain (1988)	(cited in Rieuwerts and Farago (1995)
3025-4140	Trail Canada (near Pb smelter)	Hertzman et al. (1991)	(cited in Rieuwerts and Farago (1995)
1283	Illinois (near Pb smelter)	Kimbrough et al. (1994)	(cited in Rieuwerts and Farago (1995)
114-185	Landskrona, Sweden (near Pb smelter)	Farago et al. (1999)	(cited in Rieuwerts and Farago (1995)
1984	Pribram, Czech Republic (near Pb smelter)	Rieuwerts and Farago (1996)	(cited in Rieuwerts and Farago (1995)
348	Wales, UK (near a mining site)	Gallacher et al. (1984)	(cited in Rieuwerts and Farago (1995)
340	Halkyn, UK (near a mining site)	Davies et al. (1985)	(cited in Rieuwerts and Farago (1995)
786	Shipham, UK (near a mining site)	Thornton (1988)	(cited in Rieuwerts and Farago (1995)
1870	Derbys, UK (near a mining site)	Thornton et al. (1990)	(cited in Rieuwerts and Farago (1995)
1560	Winster, UK (near a mining site)	Cotter-Howells and Thornton (1991)	(cited in Rieuwerts and Farago (1995)
726	Leadville, US (near a mining site)	Cook et al. (1993)	(cited in Rieuwerts and Farago (1995)
435	Pribram, Czech Republic (near a mining site)	Rieuwerts and Farago (1996)	(cited in Rieuwerts and Farago (1995)
857 ± 91 in PM_{60}	Jersey City, NJ	Adgate et al. (1998)	(cited in Rieuwerts and Farago (1995)
1133 ± 119 in PM_{10}	Jersey City, NJ	Adgate et al. (1998)	

Table 3-1 (cont'd). Concentrations of Lead in Indoor Dust

Concentration of Lead (ppm)	Location	Reference
975 in PM_{53}	Public school in Port Pirie, Australia	Oliver et al. (1999)
481 in PM_{250}	Public school in Port Pirie, Australia	Oliver et al. (1999)
1693-6799 in PM_{53}	Houses in Port Pirie, Australia	Oliver et al. (1999)
1407-4590 in PM_{250}	Houses in Port Pirie, Australia	Oliver et al. (1999)
558 ± 544 in TSP (airborne)	Nursing homes in Vienna	Komarnicki (2005)
612 ± 518 in PM_{10} (airborne)	Nursing homes in Vienna	Komarnicki (2005)
547 ± 512 in $PM_{2.5}$ (airborne)	Nursing homes in Vienna	Komarnicki (2005)
2.29 ($\mu g/ft^2$)	Boston, MA	cited in Lanphear et al. (1998)
293.40 ($\mu g/ft^2$)	Cincinnati, OH	cited in Lanphear et al. (1998)
20.37 ($\mu g/ft^2$)	Cincinnati, OH	cited in Lanphear et al. (1998)
8.30 ($\mu g/ft^2$)	Rochester, NY	cited in Lanphear et al. (1998)
17.79 ($\mu g/ft^2$)	Rochester, NY	cited in Lanphear et al. (1998)
2.50 ($\mu g/ft^2$)	Butte, MT	cited in Lanphear et al. (1998)
1.92 ($\mu g/ft^2$)	Bingham Creek, UT	cited in Lanphear et al. (1998)
4.73 ($\mu g/ft^2$)	Leaville, CO	cited in Lanphear et al. (1998)
8.87 ($\mu g/ft^2$)	Magna, UT	cited in Lanphear et al. (1998)
6.11 ($\mu g/ft^2$)	Sandy, UT	cited in Lanphear et al. (1998)
3.68 ($\mu g/ft^2$)	Midvale, UT	cited in Lanphear et al. (1998)
5.91 ($\mu g/ft^2$)	Palmerton, PA	cited in Lanphear et al. (1998)

3.1.2 Observed Concentrations – Outdoor

Widespread emissions from stationary and mobile sources, as well as resuspended soil, have contributed to elevated airborne lead concentrations as described in Chapter 2. In fact, airborne lead concentrations in many places throughout the world have been several orders of magnitude higher than natural background levels for the past seventy years (Miller and Friedland, 1994). The lowest concentrations measured are at the South Pole, where an average concentration of 0.076 ng/m^3 was recorded (Maenhaut et al., 1979). Even at this remote location, it is likely that the airborne lead levels are elevated above natural background. This is evidenced by lead concentrations in Antarctic snow that have risen from <1 ng/kg in 800 BC to 200 ng/kg in the 1960's (Murozumi et al., 1969).

Airborne concentrations of lead in the U.S. have fallen dramatically over the last 30 years due largely to the phase out of leaded gasoline additives. Major declines over several orders of magnitude have been observed not only in urban areas, but also in rural regions and remote locations. Figure 3-1 depicts trends in nationwide data for airborne Pb concentrations for 1985 to 2000. The national average was calculated from measurements taken at 228 monitoring sites. Data taken at rural sites throughout the United States since 1979 showed a similar decline (Eldred and Cahill, 1994). Figure 3-2 shows the overall U.S. trend for airborne lead concentrations in relation to the Lead NAAQS, from 1983 to 2002.

The United States has not been the only country to see a significant drop in lead concentrations. In the early 1980s, 5% of Europe's urban population was exposed to concentrations above the World Health Organization's recommended limit of 0.5 μg/m^3 for an annual average (Fenger, 1999; WHO, 2000). By the late 1980s, this value had fallen and there were very few locations reporting concentrations above 0.5 μg/m^3. These areas were primarily near large, uncontrolled metal industries (Fenger, 1999). Measurements made in Bermuda between 1993 and 1994 showed that despite its remote location, airborne lead concentrations had fallen by an order of magnitude since the 1970s and by a factor of four since the 1980s (Huang, 1996). Similarly, measurements taken at the South Pole were routinely below the detection limit in 2000-2001, which indicates a significant improvement in Antarctic air quality since the 1970s (Arimoto et al., 2004). Table 3-2 lists literature data on airborne lead concentrations. It should be noted that concentrations are not directly comparable due to different measurement time scales, sampling equipment, and analytical methods.

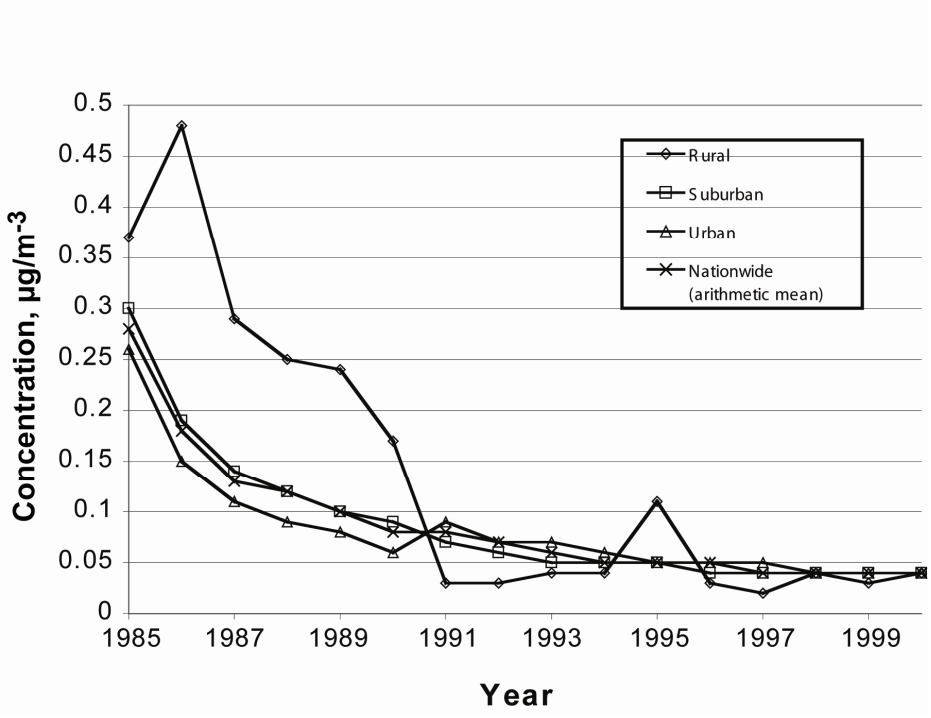


Figure 3-1. Concentrations of lead throughout the United States. The maximum quarterly averages are given for 228 urban, suburban, and rural locations.

Source: U.S. EPA (2003).

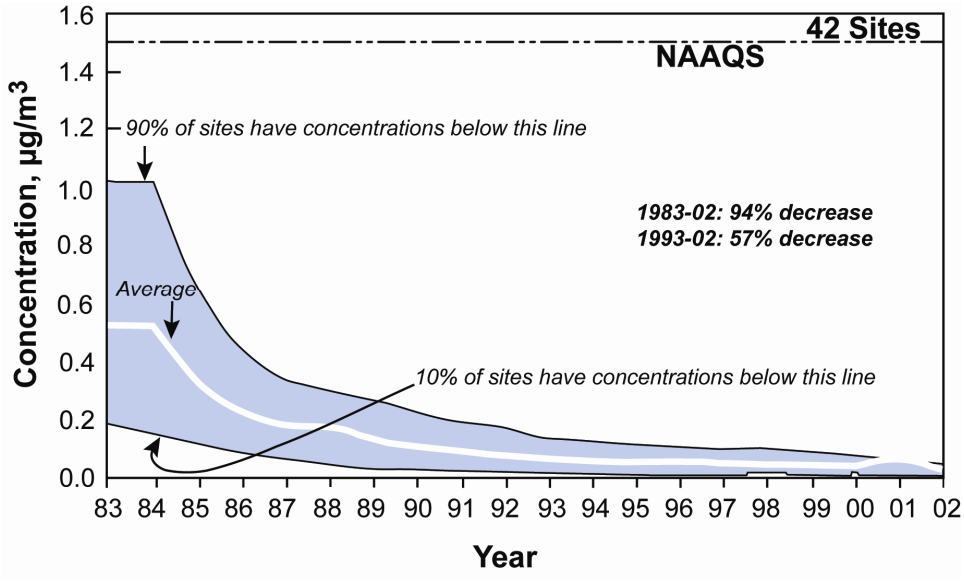


Figure 3-2. Airborne concentrations of lead, averaged across the U.S., shown in relation to the current NAAQS, for the years 1983 through 2002.

Table 3-2. Airborne Concentrations of Lead

Airborne Conc (ng/m^3)	Location	Reference
	Urban	
326 ± 15.6 in fine mode	Boston, MA	Thurston and Spengler (1985)
75.6 ± 5.95 in coarse mode	Boston, MA	Thurston and Spengler (1985)
330	Clemson, SC	Del Delumyea and Kalivretenos (1987)
52	Akron, Oh	Del Delumyea and Kalivretenos (1987)
31	Norfolk, VA	Del Delumyea and Kalivretenos (1987)
64	Chicago, IL	Del Delumyea and Kalivretenos (1987)
30-96270	Range reported in U.S. in lit	Schroeder et al. (1987)
12 ± 6	Cadiz, Spain	Torfs and Van Grieken (1997)
10 ± 8	Bari, Italy	Torfs and Van Grieken (1997)
64 ± 47	Malta, Malta	Torfs and Van Grieken (1997)
110 ± 65	Eleusis, Greece	Torfs and Van Grieken (1997)
4-444	Caesarea, Israel	Erel et al. (1997)
45 ± 16	Geneva, Switzerland	Chiaradia and Cupelin (2000)
49 ± 43	Vancouver, BC	Brewer and Belzer (2001)
13.1	Riverside, CA	Hui (2002)
15.4-18.9	Los Angeles, CA	Hui (2002)
6.9	San Francisco, CA	Hui (2002)
22 ± 17	Jerusalem, Israel	Erel et al. (2002)
<40	Yerevan, Armenia	Kurkjian et al. (2002)
230-650	St. Louis, MO	Kim et al. (2005)
	Rural	
16	Packwood, WA	Davidson et al. (1985)
2-1700	Range reported in U.S. in lit	Schroeder et al (1987)
9	Whiteface Mountain, NY	Miller and Friedland (1994)
2.5	IMPROVE network	Eldred and Cahill (1994)
0.54-6.34	IMPROVE network	Malm and Sisler (2000)
28.6	Lake Balaton, Hungary	Hlavay et al. (2001)
	Remote	
2.2	Olympic National Park	Davidson et al. (1985)
4.6	Glacier National Park	Davidson et al. (1985)
15	Great Smoky Mt. National Park	Davidson et al. (1985)
0.007-64	Range reported in lit	Schroeder et al (1987)
0.04-3.2	Bermuda	Huang et al. (1996)
<0.032	Antarctica	Arimoto et al. (2004)

Table 3-2 (cont'd). Airborne Concentrations of Lead

Airborne Conc (ng/m^3)	Location	Reference
	Near Sources of Lead Emissions	
1700-4000	Fenceline of a lead smelter, CA, downwind	Kimbrough and Suffett (1995)
960-1200	Fenceline of a lead smelter, CA, upwind	Kimbrough and Suffett (1995)
758	Jerusalem-Tel Aviv freeway, Israel	Erel et al. (1997)
400-1000	Australia roadsides	Al-Chalabi and Hawker (1997)
127-173	Hong Kong roadsides	Chan et al. (2000)
46-113	Gothenburg, Sweden roadsides	Sternbeck et al. (2002)
27.4	Birmingham, UK roadside	Harrison et al. (2003)

Concentrations of airborne lead are sometimes several orders of magnitude higher in urban areas compared to remote regions (Schroeder et al., 1987; Malm and Sisler, 2000). Rural areas tend to have concentrations falling somewhere between those of urban and remote areas. Urban populations, which tend to be comprised of low-income peoples and/or minorities, are exposed to comparatively higher levels of airborne lead.

A substantial amount of data for airborne lead outside of urban areas is available from the Interagency Monitoring of Protected Visual Environments Network, known as IMPROVE. This network started in 1987 and includes 110 sites nationwide (California Air Resources Board, 2000). The sites are located at rural and remote sites that are impacted by transport of fine particles which reduce visibility. The network uses a sampling frequency of once every three days, and provides $PM_{2.5}$ lead concentration data determined by Proton-Induced X-ray Emission.

The highest lead concentrations reported by the IMPROVE Network are found in southern California and much of the eastern United States (Malm and Sisler, 2000). Data indicate that high concentrations of lead in the west are found in the urban corridor along the Cascade Mountains in Washington, Oregon, and Utah at around 5.0 ng/m^3 (Malm and Sisler, 2000). Rural concentrations in the eastern U.S. are typically in the range of 6-10 ng/m^3. Airborne concentrations throughout the United States outside of cities are shown on an isopleth map in Figure 3-3.

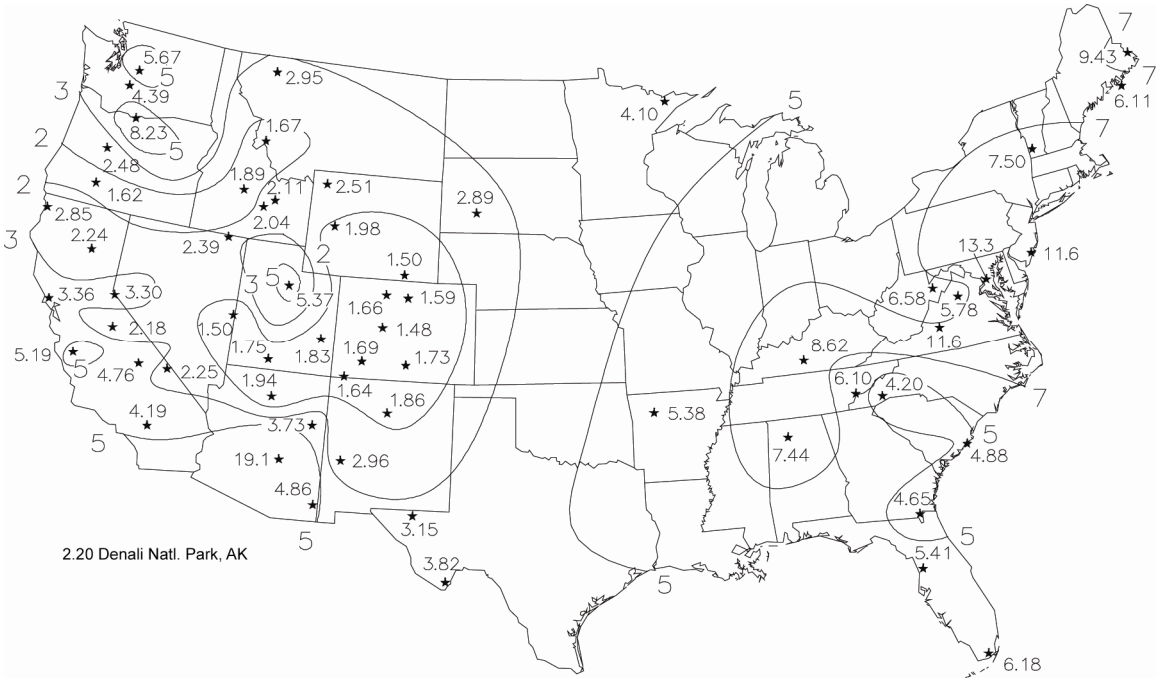


Figure 3-3. Concentrations of lead measured in 1995-1998 as recorded by the IMPROVE network. Data are given in ng/m^3. (Reprinted from Malm and Sisler, 2000).

The quarterly average airborne concentration of lead cannot exceed 1.5 μg/m^3, according to the 1978 air quality standard. Between September 2001 and September 2002, there were just four areas in the United States that were not in attainment of this standard: Liberty-Acadia, MO, Herculaneum, MO, East Helena, MT, and Lame Deer, MT (U.S. Environmental Protection Agency, 2003). As of 2004, there were only two areas out of attainment (www.epa.gov/air/oaqps/greenbk/inte.html).

Some seasonal variability is common for lead concentrations. However, whether seasonal variability is present depends on precipitation trends, changes in wind direction, and mixing height variability for a given area. A relative maximum was observed in the winter in the Arctic because of the lack of precipitation during winter months (Heidam, 1986). However, in Bermuda a relative maximum was observed in the summer when winds come predominantly from Africa and Europe (Huang, 1996). Chiaradia and Cupelin (2000) observed no seasonality in lead concentrations in Geneva, Switzerland. Measurements taken at a number of U.S. and

French cities suggest some variation based on seasonal differences in mixing height (Delumyea and Kalivretenos, 1987).

Measurements made in Riverside, CA after lead was removed from gasoline show diurnal trends (Singh et al., 2002). Concentrations of lead are high in the morning (6am – 10am) and the late afternoon (4 pm – 8 pm). This is probably indicative of heavy traffic, a depressed atmospheric mixing height in the morning, and advection from Los Angeles traffic. Lead concentrations in Riverside are significantly lower during midday (10am – 4pm) and night (8 pm – 6 am).

Concentrations of lead are dependent on height. This is particularly true if lead is emitted at street level from traffic. Measurements performed at roadsides in Hong Kong in 1997 show much higher concentrations at breathing level than at rooftop level (Chan et al., 2000). Similarly, lead concentrations measured at four elevations in Berne, Switzerland show a pronounced decrease with height (Gaelli and Nyffeler, 1987). Some leaded gasoline was still used in Hong Kong and Switzerland during these two studies. Measurements made in an urban street canyon in Lahti, Finland show that concentrations declined by a factor of five between street level (1.5m) and rooftop level (25m) (Väkevä et al., 1999).

3.1.3 Observed Concentrations – Occupational

Lead concentrations inside work places can also be elevated. Thus, inhalation of lead during work hours is an additional route of exposure for some subpopulations.

Feng and Barratt (1994) measured concentrations of lead in two office buildings in the UK. In general, concentrations in the office buildings were higher than concentrations in nearby houses. Office dust was concentrated in the organic and residual fractions unlike house dust which was bound to carbonate and Fe-Mn oxides. This indicates that offices and houses may have different lead sources. Office building lead also tends to be in the coarse mode, unlike house dust that is predominantly in fine particles (Feng and Barratt, 1994).

As expected, concentrations of lead tend to be highly elevated within manufacturing facilities for lead-based products (Rieuwerts et al., 1999; Harrison et al., 1981; Tsai et al., 1997). This is a major exposure route for employees. Measurements taken in a battery manufacturing plant show lead concentrations in floor dust to be 47,700 ppm outside of the assembly plant,

39,200 ppm inside the assembly plant, and 73,700 ppm in the battery grid storage area (Rieuwerts et al., 1999).

Airborne concentrations of lead in a battery manufacturing plant, a metallic film capacitor plant, and a lead powder plant were 140 ± 112 $\mu g/m^3$, 281 ± 114 $\mu g/m^3$, and 485 ± 245 $\mu g/m^3$ respectively (Tsai et al., 1997). Work sites that use mechanical actions such as abrasion, friction, and cutting typically generate large particles. However, work sites that use high temperature operations generate small, respirable particles. At the three sites listed above, particle sizes were predominantly >10 μm in diameter (Tsai et al., 1997).

A Pb-Zn smelter in the UK similarly showed much larger lead particle sizes inside the facility than outside of the facility (Harrison et al., 1981). This may be because concentrations are high enough indoors to coagulate. Floor dusts (<60 μm) taken from each process site in the overall smelting process contained the same lead species as the aerosols emitted from each process, which are discussed in Section 2.2.

Residential renovation and paint removal are major sources of lead exposure for both workers and residents. Dry sanding, abrasive blasting, and burning, welding, or heating surfaces generate highly dangerous levels of lead (Jacobs, 1998). The geometric mean and maximum concentrations during each of these processes are listed in Table 3-3. Daniels et al. (2001) measured airborne concentrations of lead during exterior paint removal from residences via wet abrasive blasting technology. The eight-hour, time-weighted average (TWA) exposures measured through personal monitors ranged between 55.1 and 81.5 $\mu g/m^3$. Area concentrations were between 20.5 and 26.9 $\mu g/m^3$.

Lead-based paints were the predominant coating for highway bridges for many years. Paint removal during renovation projects has also been cited as a major source of lead exposure for workers. As with residential renovation, lead concentrations during industrial paint removal depend largely on the technology used. Generally, abrasive blasting techniques are used, which breaks lead coatings into small particles that can be inhaled or ingested if hands are not washed prior to eating or smoking (Chute and Mostaghim, 1991). Vacuum blasting may reduce occupational exposures. Personal monitors worn during vacuum blasting on a bridge registered concentrations between 27 and 76 $\mu g/m^3$ with a geometric mean of 55 $\mu g/m^3$ (Mickelson and Johnston, 1995). Concentrations fell to 0.1 and 2 $\mu g/m^3$ over an eight-hour TWA eleven meters from the removal processes.

Table 3-3. Airborne Concentrations Surrounding Residential Lead-Based Paint Abatement

Abatement Technique	Geometric Mean ($\mu g/m^3$)	Maximum Exposure ($\mu g/m^3$)
Preparation (e.g., carpet removal)	2	206
Abrasion	8	403
Chemical stripping	3	476
Encapsulation	2	72
Heat gun	7	915
Component replacement	3	121
Cleaning	2	590

Source: Jacobs (1998).

Lead concentrations measured in underground gold mines were somewhat elevated, but comparable to ambient concentrations due to adequate air exchange. Measurements made in a gold mine in South Africa were highly dependent on the process being undertaken (Annegarn et al., 1988). Concentrations ranged between 1.4 $\mu g/m^3$ and 800 $\mu g/m^3$. A source apportionment study in a Nevada gold mine measured lead concentrations that averaged 0.21 $\mu g/m^3$ (McDonald et al., 2003).

3.2 EXPOSURE: SOIL AND ROAD DUST

Contaminated soil can be a potential source of lead exposure for humans. Soil lead can be directly ingested through hand-to-mouth behavior common in children, indirectly ingested through contaminated food, or inhaled when breathing air containing resuspended soil particles.

Here we address soil concentrations measured in urban, residential, and industrial areas. Soil lead concentrations in rural and remote areas, agricultural soils, and sediment are addressed in Chapter 8 of this document.

The natural background concentration of lead in soil is estimated to be in the range of 1–200 ppm with an average of 15 ppm (Zimdahl and Skogerboe, 1977). It should be noted that soil lead measurements are difficult to compare given the variety of extraction techniques and depths of soil cores analyzed in each study.

The dominant source of lead to soil is atmospheric deposition both from local sources and long-range transport. In general soil in urban and residential areas is contaminated primarily via atmospheric deposition, direct application of agricultural chemicals, and natural mineral weathering of parent rock (Paces, 1998). At a local level, soil lead contamination can be derived from agricultural and food wastes, animal wastes and manure, logging and other wood-cutting activities, urban refuse, municipal sewage sludge, miscellaneous organic wastes including excreta, solid wastes from metal manufacturing, coal fly ash and bottom fly ash, peat for agricultural and fuel uses, wastage of commercial products, mine tailings, and smelter slags and wastes (Nriagu and Pacyna, 1988). Flaking and peeling of lead-based paint can also be a significant source of soil lead near old structures (Small et al., 1995; Finkelstein et al., 2003).

Soil Response Times

The retention time for lead in the soil is much longer than it is in the air. The only "removal" mechanisms for soil lead are resuspension and leaching, the latter of which is known to be a slow process (see Chapter 2 of this document for details). The retention time, or the amount of time required to reduce the soil concentration by half, is estimated to be on the order of hundreds to thousands of years (Dudka and Adriano, 1997). Box model estimates based on data for an agricultural catchment in the Czech Republic predict that steady state concentrations for soil lead will not be achieved for 980 years (Paces, 1998). Modeling efforts by Harris and Davidson (2005) in southern California similarly predict that steady state concentrations of soil lead will not be achieved for hundreds of years assuming emissions rates stay constant. The lowest estimates of a response time are given by Miller and Friedland (1994) in the northeastern United States. They estimate that soil lead concentrations in a northern hardwood forest zone will stabilize in just 17 years and soil lead concentrations in a subalpine spruce-fir forest zone will stabilize in 77 years. A later study in the same region estimated the response times as 60 years and 150 years for the two forests, respectively (Kaste et al., 2003).

3.2.1 Urban Background Concentrations of Soil Lead

The concentration of soil lead varies significantly throughout urban areas depending on proximity to stationary sources and roadways, and wind direction and speed.

The major source of lead in urban soils is automotive traffic (Sheets et al., 2001; Mielke, 1993; Sutherland, 2000). Lead remains in the soil for a long time. Thus, much of the existing urban soil is likely a remnant of leaded gasoline use. Soil concentrations decrease both with depth and distance from roadways. Furthermore, in several urban areas there was little correlation between soil lead and the age of nearby houses, which suggests that lead-based paint is not as significant of a source as automotive lead (Mielke, 1993).

The concentrations of lead in soil depend primarily on the size of the city and the location within the city (Mielke, 1991, 1993). Extensive lead studies performed in Baltimore, New Orleans, and cities throughout Minnesota demonstrate that the highest concentrations of lead are found in the central sections of the city where traffic and population density are greatest (Mielke, 1991, 1993). The lowest concentrations are found in the outskirts of these cities and in smaller cities. In all of these studies, the age of housing does not seem to be a major factor, which suggests that the impacts of lead-based paint are dominated by historic emissions of leaded gasoline additives. That the highest concentrations are in the inner city, generally populated by minorities and the poor, suggests that they are the groups most at risk for lead exposure from contaminated soil.

The highest concentrations are observed near major roadways. Surface soil lead concentrations measured near a major freeway in Cincinnati, OH, for example, are between 59 ppm and 1980 ppm, which is well above background (Turer et al., 2001). These concentrations drop off dramatically with depth. An estimated 40% of lead from exhaust is retained in the nearby soil (Turer et al., 2001).

Measurements of Erel et al. (1997) in Israel show that soil lead concentrations decrease more rapidly with depth near roadways than far from roadways. In a soil profile extracted near a local road, lead concentrations fell by a factor of 42 between the surface and 30-36cm from the surface. However, far from the roadway, lead concentrations fell by about a factor of 3 between the surface and 30-36cm below the surface.

Several authors making measurements during the days of leaded gasoline usage reported elevated lead concentrations in soil that decrease with distance from roadways. For example, Pierson and Brachaczek (1976) report soil lead levels that decrease from >1000 ppm adjacent to the road down to less than 200 ppm at 12.5 m from the roadway edge. Harris and Davidson (2005) have shown through a mass balance model that elevated lead concentrations in soil are

likely to remain high for hundreds of years; this is consistent with other studies showing similarly long residence times in soil (e.g., Dudka and Adriano, 1997).

Soil lead concentrations in urban areas are generally higher than soil lead concentrations in rural or remote areas. The average concentrations of soil lead in urban areas are shown in Table 3-4. In many cases these data are averages over commercial, residential, and public areas, which cover a wide range of concentrations.

Table 3-4. Concentration of Soil Lead in Urban Areas

Location	Soil conc. (ppm)	Depth (cm)	Reference
Springfield, MO	107 ± 8	0-15	Sheets et al., 2001
Urban locations throughout Egypt	23-200	0-30	Badawy et al., 2002
southern California	65.2, 66.3, 99.4	0-10	Young et al., 2002
central New Orleans, LA	4-69000	0-2.5	Mielke, 1993
outer New Orleans, LA	1-24400	0-2.5	Mielke, 1993
suburban New Orleans, LA	2-5650	0-2.5	Mielke, 1993
Baton Rouge, LA	2-6680	0-2.5	Mielke, 1993
Monroe, LA	8-11600	0-2.5	Mielke, 1993
Alexandria, LA	6-2590	0-2.5	Mielke, 1993
Lafayette, LA	6-8860	0-2.5	Mielke, 1993
Natchitoches, LA	6-1430	0-2.5	Mielke, 1993
Reno-Sparks, NV	~10	0-1	Gillies et al., 1999
Manoa, Hawaii	58 ± 27	0-2.5	Sutherland, 2000
Gainesville, FL	~16	0-20	Chirenje et al., 2004
Miami, FL	~93	0-10	Chirenje et al., 2004

3.2.2 Soil Concentrations Near Stationary Sources

Concentrations Near Lead Smelters

Lead in soil is highly elevated near sources of lead emissions. In particular, stationary facilities such as smelters and battery disposal sites can have very high levels of soil lead.

Major smelter deposits exist primarily within a 0.5 km radius of the stack (Chatterjee and Banerjee, 1999; Rieuwerts et al., 1999) although some studies observe elevated concentrations of lead as far away as 30 km (Liu, 2003). Franssens et al. (2004) used isotopic measurements to show that between 50% and 80% of dry depositing lead within a 3–4 km radius of a lead-zinc smelter had an industrial origin.

Soil concentrations of lead decrease dramatically with distance from the source and depend greatly on windspeed and direction (Kimbrough and Suffet, 1995; Palacios et al., 2002; Suchara and Sucharová, 2004). Godin et al. (1985) measured soil concentrations that were

almost proportional to the inverse of the distance from the source and the square root of the wind frequency. Suchara and Sucharová (2004) estimate an exponential decrease in soil concentration with distance from a lead smelter in the Czech Republic. Data collected within a 14 km radius showed an exponential decrease in soil lead concentration with distance from the source. Exponential decreases in soil concentrations have been suggested elsewhere as well (e.g., Chatterjee and Banerjee, 1999; Rieuwerts et al., 1999). Examples of data showing decreases in soil concentration with distance from major sources are shown in Table 3-5.

Table 3-5. Concentrations of Soil Lead with Distance from Lead Smelters

Distance from Smelter (m)	Concentration (ppm, dry weight)		
fenceline	2300[a,4]	46700 ± 2100[a,5]	12650[b,6]
20	5657[d,1]		
30	3937[d,1]		
40	3253[d,1]		
100	783[d,1]	312.8 ± 98.7[e,2]	1800[a,4]
123 - 256	636 ± 522[c,8]		
250	229[d,1]	20200 ± 1100[a,5]	
400	127[d,1]		
500	400 ± 20[a,5]		
700	792[e,7]		
1500	519[c,3]		
3000	242[c,3]		
5000	216.7 ± 87.6[e,2]	137[c,3]	
10000	110.3 ± 76.4[e,2]		
20000	57.4 ± 24.9[e,2]		
30000	32.9 ± 21.4[e,2]		

Note: In cases where multiple transects were sampled, only the downwind transects are shown.

[a]Depth sampled was not defined
[b]Sample depth was 0-5 cm
[c]Sample depth was 0-10 cm
[d]Sample depth was 0-15 cm
[e]Sample depth was 0-30 cm

[1]Palacios et al. (2002)
[2]Liu (2003)
[3]Godin et al. (1985)
[4]Kimbrough and Suffet (1995)
[5]Chatterjee and Banerjee (1999)
[6]Rieuwerts et al. (1999)
[7]Venditti et al. (2000)
[8]Young et al. (2002)

As is the case with urban soils, lead concentrations decrease significantly with depth near industrial sites. Results of Chatterjee and Banerjee (1999) indicate that lead concentrations remain relatively constant within about 250 meters and decrease with distance after this. Table 3-6 lists a lead concentration profile measured near a lead smelter in northern France.

Table 3-6. Soil Lead Concentration Profile Measured Near a Lead Smelter in Northern France

Depth (cm)	Soil Horizon	Soil Conc. (ppm)
0-6	Oi	2340
6-9	Oa	4480
9-36	Ag	383
36-50	ABg	21.7
50-70	BAg	18.2
70-85	Bg	17.1
85-120	IIC2g	12.4
120-165	IIC3g	10.2

Source: Denaix et al. (2001).

The species of metals found near smelters vary depending on soil conditions. One study observed lead in topsoil that was either in the form $Pb_5(PO_4)_3Cl$ or Pb(II) compounds that were adsorbed onto Fe(II) oxides or associated with clay particles (Batonneau et al., 2004).

Other measurements at a site contaminated with automotive battery wastes showed lead species in the soil to be $Pb(CO)_3$, $Pb(CO_3)_2$, $Pb(OH)_2$, PbO, and $PbSO_4$ (Pichtel et al., 2000). Additional studies have shown lead contamination bonded to bacteria (Denaix et al., 2001), carbonate (Maskall and Thornton, 1998; Pichtel et al., 2000; Venditti et al., 2000), sulfide phases (Pichtel et al., 2000; Venditti et al., 2000), organic phases (Pichtel et al., 2000; Venditti et al., 2000) and Fe-Mn oxides (Venditti et al., 2000). The prevalence of carbonate forms in contaminated soil is due to coinciding contamination with calcareous slag wastes (Maskall and Thornton, 1998).

Lead concentrations do not appear to have decreased surrounding smelters despite the implementation of pollution controls. A smelter in Slovenia was fitted with protective filters in 1978 (Zadnik, 2004). Since that time concentrations have fallen dramatically in hay samples and cow blood within 10 km of the smelter however, soil concentrations did not decrease between 1978 and 2003 (Zadnik, 2004). Similarly a lead-zinc smelter in British Columbia, Canada was replaced by a new smelting facility in 1997 (Hilts, 2003). Lead concentrations fell by 50% in outdoor dustfall, street dust, and indoor dustfall. Airborne concentrations fell by nearly 75%. However, no statistically significant decline was observed in soil or carpet concentrations (Hilts, 2003).

Lead in soil appears to be associated with relatively small soil particle sizes. Young et al. (2002) observed that the lead concentration was much higher in the <38 μm size range than in the 300 μm-2mm size range in contaminated soils. This is likely due to the higher specific surface area of smaller soil particles and that lead tends to bond with organic matter and Fe/Al oxides, which can also concentrate in smaller size particles. (Young et al., 2002).

Concentrations Near Mines

Concentrations of lead are highly elevated near mines as well. Lead and zinc mines in particular have large deposits of lead in nearby soil, but mines used for extracting other metals can also have lead contaminated soil. Mine sites are contaminated by the disposal of mine tailings, acid mine drainage, and atmospheric deposition of airborne emissions (Dudka and Adriano, 1997). Mines in the United States produced an estimated 480 Tg of lead tailings and 50 Tg of lead mine wastes between 1910 and 1981 (Dudka and Adriano, 1997).

Lead is widely dispersed surrounding mining sites (Dudka and Adriano, 1997; Rieuwerts and Farago, 1995). Thus, it is not easy to determine a relationship between distance and soil concentration as is the case with smelting emissions. However, a study of an abandoned lead-zinc mine in Tyndrum, Scotland located near a river showed that fluvial transport had carried lead contamination at least as far as 6.5 km although contamination is suspected as far as 25 km downstream (MacKenzie and Pulford, 2002). Soil concentrations measured near mining sites are shown in Table 3-7.

Table 3-7. Soil Concentrations Measured Near Mining Sites

Location	Type of Mine	Main Period of Operation	Depth (cm)	Mean conc. (ppm)	Reference
Wales, UK	Pb	historic, not specified	0–15	1159	Gallacher et al. (1984) (taken from Rieuwerts and Farago, 1995)
Halkyn, UK	Pb-Zn	1845–1938	0–15	1127	Davies et al. (1985) (taken from Rieuwerts and Farago, 1995)
Shipham, UK	Zn, Pb	1700–1850	0–15	7900	Mattigod et al. (1986) (taken from Rieuwerts and Farago, 1995)
Shipham, UK	Zn, Pb	1650–1850	0–5	2002	Thornton (1988) (taken from Rieuwerts and Farago, 1995)
Derbys, UK	Pb	18th and 19th cent.	0–5	5610	Thornton (1990) (taken from Rieuwerts and Farago, 1995
Winster, UK	Pb	Up to end of 18th cent.	0–5	7140	Cotter-Howells and Thornton (1991) (taken from Rieuwerts and Farago, 1995)
Leadville, US	Pb	1860s–1960s	n.a.	1110	Cook et al. (1993) (taken from Rieuwerts and Farago, 1995
Derbys, UK	Pb	18th and 19th cent.	0–15	1800	Li and Thornton (1993) (taken from Rieuwerts and Farago, 1995)
Shipham, UK	Zn, Pb	18th and 19th cent.	0–15	7360 (max)	Li and Thornton (1993) (taken from Rieuwerts and Farago, 1995)
Pribram, Czech Republic	Pb	18th–20th cent.	0–5	1451	Rieuwerts and Farago (1996) (taken from Rieuwerts and Farago, 1995)
Tyndrum, Scotland	Pb-Zn	Up to 1862	n.a.	13000	MacKenzie and Pulford (2002)
Goldenville, Canada	Au	1869–1927	n.a.	70–120	Wong et al. (2002)
São Domingos, Portugal	Cu	Pre-Roman–Roman times	0–30	2694	Freitas et al. (2004)
Dubuque, U.S.	Zn, Pb	19th century	0–20	791	Mbila and Thompson (2004)

Lead is found in many different forms near mining sites. It is commonly found in its mineral form of galena (Rieuwerts and Farago, 1995; Dudka and Adriano, 1997). However, in mine spoils, lead is also found as plumbojarosite [$PbFe_6(SO_4)_4(OH)_{12}$], pyromorphite [$Pb_5(PO_4)_3Cl$], lead carbonate [$PbCO_3$], leadhillite [$Pb_4SO_4(CO_3)_2(OH)_2$], $PbS•Bi_2S_3$, lead

oxides, lead silicates, and lead sulfate [$PbSO_4$] (Rieuwerts and Farago, 1995; Mbila and Thompson, 2004).

Lead tends to be more heavily concentrated in smaller soil grain sizes than in larger grain sizes (MacKenzie and Pulford, 2002). Results of one study are listed in Table 3-8. Additionally, Rieuwerts and Farago (1995) note that soil lead particles are typically larger in mining areas than in smelting areas.

Table 3-8. Concentrations of Lead in Soils Grouped by Soil Grain Size

Size Fraction	Pb conc. of main mine waste	Pb conc. of processing site waste
>180 µm	0.91%	17%
53-180 µm	1.5%	14%
<53 µm	4.5%	18%

Source: MacKenzie and Pulford (2002).

In addition to soil, lead concentrations in peat have been shown to decrease with depth. Figure 3-4 illustrates two peat profiles sampled near an abandoned lead mine.

3.2.3 Concentrations of Lead in Road Dust

Elevated concentrations of lead in road dust pose an important exposure risk through wind and traffic resuspension as outlined in Chapter 2 of this document.

The primary source of lead in road dust is adjacent soil (de Miguel et al., 1997). However traffic emissions, the weathering and corrosion of building materials (de Miguel et al., 1997), and brake pad wear (Garg et al., 2000) are additional sources. Between 60–90% of the mass of road dust is comprised of soil particles (Adgate et al., 1998). As mentioned above soil is still an important reservoir for lead emitted from vehicles despite the widespread phase out of leaded gasoline.

The concentration of lead in road dust is generally elevated above background. This is particularly true in urban areas. Additionally, measurements reported in 2003 in the San Joaquin Valley of California show concentrations that are significantly lower than concentrations

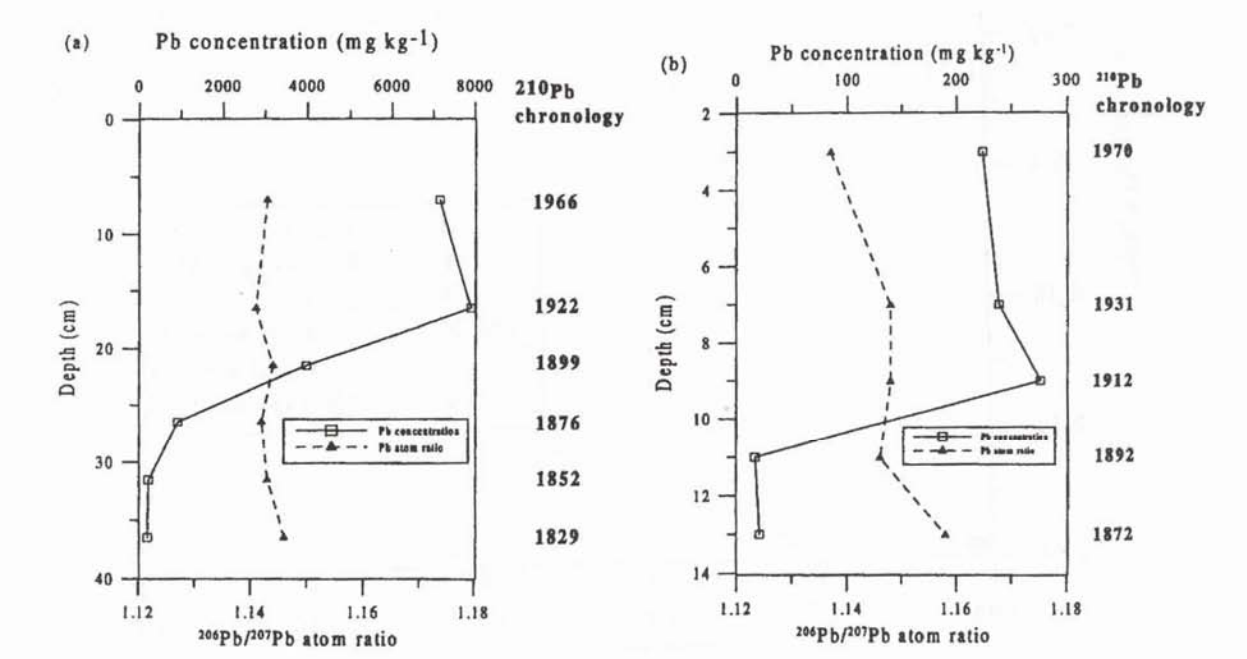


Figure 3-4. The changes in lead concentration with depth in two peat cores. Core A was taken at a location adjacent to the ore processing area of the abandoned lead mine in Tyndrum, Scotland. Core B was taken 0.5 km from the main mine waste dump at the same site.

Source: MacKenzie and Pulford (2002).

measured in the same area in 1987 (Chow et al., 2003). Lead data reported in the literature are listed in Table 3-9.

Metals in road dust tend to be associated with small size grains. Measurements of Kuang et al. (2004) show that metals are concentrated in grains smaller than 0.125 mm in diameter.

De Miguel et al. (1997) observe a steep gradient in road dust concentrations of lead in the north-south direction in Oslo, Norway. This indicates that lead concentrations are much higher in the highly urbanized areas and lower in the suburban and residential areas. This is consistent with traffic and building construction, renovation, and weathering of building materials being the dominant source of lead to soil and subsequently road dust (de Miguel et al., 1997).

Table 3-9. The Concentration of Lead in Road Dusts

Conc. of Lead (ppm)	Location	Land Use	Reference
180 ± 14	Oslo, Norway	urban, paved road	de Miguel et al., 1997
1927 ± 508	Madrid, Spain	urban, paved road	de Miguel et al., 1997
536 ± 39	Calcutta, India	near lead smelter, paved	Chatterjee and Banerjee, 1999
57.2 ± 27.3	Beijing, Chinga	urban, paved road	Kuang et al., 2004
~100	Reno-Sparks, NV	urban, paved road	Gillies et al., 1999
1209 ± 170 (PM2.5)	Hong Kong	urban, paved road	Ho et al., 2003
1061 ± 155 (PM10)	Hong Kong	urban, paved road	Ho et al., 2003
588 ± 688	Honolulu, HI	urban, paved road	Sutherland et al., 2003
470 ± 524	Honolulu, HI	urban, paved road	Sutherland et al., 2003
151 ± 124	Honolulu, HI	urban, paved road	Sutherland et al., 2003
161 ± 31	San Joaquin Valley, CA	urban, paved road	Chow et al., 2003
57 ± 28	San Joaquin Valley, CA	rural, paved road	Chow et al., 2003
109 ± 74	San Joaquin Valley, CA	composite, paved road	Chow et al., 2003
58 ± 73	San Joaquin Valley, CA	agricultural unpaved road	Chow et al., 2003
203 ± 133	San Joaquin Valley, CA	residential unpaved road	Chow et al., 2003
43 ± 8	San Joaquin Valley, CA	staging area soil	Chow et al., 2003
101 ± 88	San Joaquin Valley, CA	unpaved composite	Chow et al., 2003

3.3 EXPOSURE: DRINKING WATER

Lead in drinking water is primarily a result of corrosion from lead pipes, lead-based solder, or brass or bronze fixtures within a residence (Lee et al., 1989; Singley, 1994; Isaac et al., 1997). Very little lead in drinking water comes from utility supplies. Experiments of Gulson et al. (1994) confirm this by using isotopic analysis. Tap water analyzed in a public school, apartments, and free standing houses also indicates that the indoor plumbing is a greater source of lead in drinking water, even for residences and schools serviced by lead-pipe water mains (Moir et al., 1996). Ratios of influent lead concentration to tap concentrations in homes in four municipalities in Massachusetts ranged between 0.17 to 0.69, providing further confirmation that in-home lead corrosion dominates the trace quantities of lead in municipal water supplies (Isaac et al., 1997).

The material of pipes is of great importance when considering how much lead is leached into drinking water. Copper piping with lead-based solder has largely replaced pure lead piping in the United States. A survey of 94 water companies nationwide in 1988 revealed that copper pipe was present in 73% of homes, galvanized pipe was present in 13% of homes, a mixture of galvanized and copper was present in 11% of homes, and plastic pipes were present in 2% of homes (Lee et al., 1989). An analysis of PVC pipes indicated that lead leached from PVC in measurable amounts,(Sadiq et al., 1997). PVC, which contains ~1% lead, increased the tap concentration to an average of 0.017 ± 0.038 mg/L, which is a statistically significant increase

over the influent concentration of 0.011 ± 0.026 mg/L (Sadiq et al., 1997). Guo et al. (1997) suggested that lead may be leached from cement-mortar lined pipes in significant quantities if the cement was made from clinker derived from combusted, hazardous materials.

In addition to piping, lead may leach from faucets. Measurements performed on 12 faucets of different compositions typically found in homes indicated that new cast-brass faucets leached more lead than any of the other designs (Gardels and Sorg, 1989). Lead levels were below the detection limit from a plastic faucet. In houses with copper piping and lead-based solder, brass fixtures may contribute as much as 50% of lead in drinking water (Lee et al., 1989).

The primary type of solder used in the U.S. is 50–50 tin-lead solder (50% tin, 50% lead). In comparing lead leached from 50–50 tin-lead solder, 95–5 tin-antimony solder, and a liquefied 50–50 tin-lead formulation that contained a flux, Birden et al. (1985) showed that the liquefied 50–50 formulation leached the most lead into drinking water. The 9–5 tin-antimony solder was the safest with respect to drinking water quality. Measurements of metals leached from four, nonlead-based solders in copper pipes were undertaken by Subramanian et al. (1991, 1994). Of the four solders tested (95–5 Sn-Sb, 96–4 Sn-Ag, 94–6 Sn-Ag, and 95.5–4.0-0.5 Sn-Cu-Ag) all showed that metals (Ag, Cd, Cu, Sb, Sn, and Zn) were leached in small enough quantities to make these solders safe alternatives to lead-based solders.

Lead corrosion is essentially an electrochemical process. Electrons may be transferred from the metal (lead) to the solution (drinking water) where the major electron acceptors are dissolved oxygen, hydrogen ions, or disinfectant residuals (Singley, 1994). Alternatively, when two different metals are in contact, there is a difference in potential and the difference in electron demand may increase corrosion (Singley, 1994). In either case, lowering the pH and increasing the dissolved oxygen demand are known to increase rates of corrosion. The corrosion process occurs faster at high temperatures than at low temperatures (e.g., Thompson and Sosnin, 1985; Lee et al., 1989).

The combined pH and alkalinity of water are sometimes described as the aggressiveness of the water and is measured using the Langelier Index. A pH above 8.0 is generally considered safe for lead leaching (e.g., Lee et al., 1989; Frey, 1989).

There are conflicting reports on the effect of chlorine in water. Chlorine, which is typically used as a disinfectant in municipal supplies, may increase the rate of corrosion by

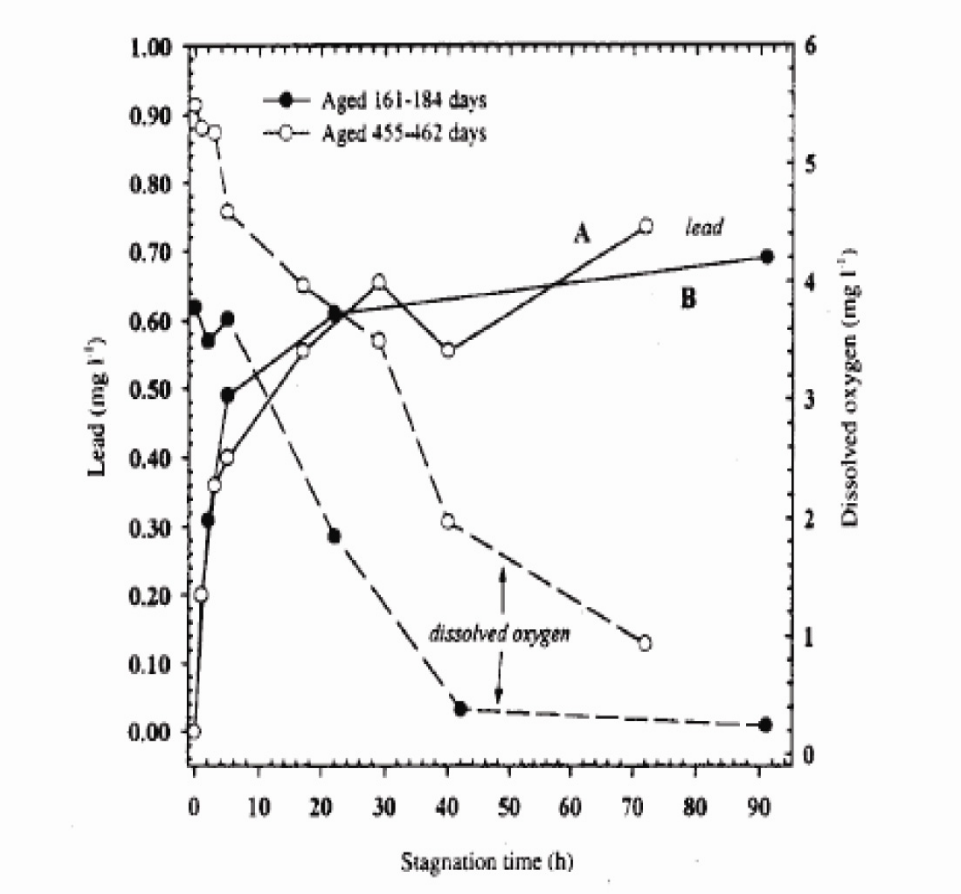
providing a source of electron acceptors (Singley, 1994). However, measurements of Lee et al. (1989) show an absence of statistically significant change in lead levels with increasing concentration of free chlorine. Laboratory tests of Edwards and Dudi (2004) show that chlorine reacts with soluble Pb^{2+} to precipitate a red-brown colored lead solid. This solid is highly insoluble, even at a pH of 1.9 for twelve weeks. Thus, chlorine may actually lessen the overall quantity of lead in drinking water. Elevated levels of lead in drinking water in Washington DC in 2000 were traced to a change from chlorine to chloramine disinfectant. The red-brown lead solid does not form in the presence of chloramines, and the data suggest that chloramines dramatically increase the amount of lead leached from brass (Edwards and Dudi, 2004).

Flouridating water does not seem to affect the solubility or reactivity of lead compounds (Urbansky and Schock, 2000).

Corrosion inhibitors are sometimes added to water to inhibit scaling or iron precipitation. Zinc orthophosphate in the range of 0.4-0.6 mg/L is an effective inhibitor for lead corrosion (Lee et al., 1989). Results indicate that zinc orthophosphate is more effective at reducing lead levels than increasing the pH. Soluble lead release is reduced by up to 70% with the addition of orthophosphate (Edwards and McNeill, 2002). Other proposed corrosion inhibitors such as sodium zinc hexametaphosphate or sodium hexametaphosphate are not effective at reducing lead corrosion (Lee et al., 1989). In fact, results of McNeill and Edwards (2004) indicate that hexametaphosphate increased the levels of soluble lead in drinking water. Each milligram per liter of hexametaphosphate increased the lead content by ~1.6 mg/L after a 72 hour stagnation period in pure lead pipes (Edwards and McNeill, 2002).

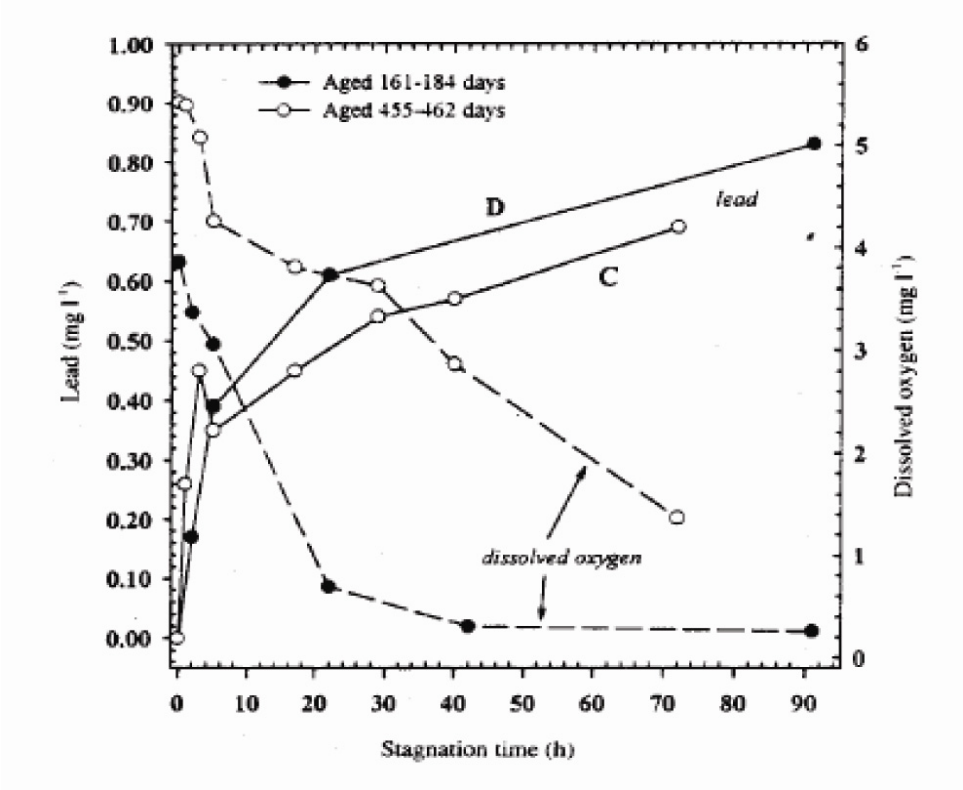
The length of time that drinking water remains in a pipe also affects the lead concentration. Thus, a first flush phenomenon is generally observed in the morning after water has stayed in the pipe through the night. An estimated 47% of total leached lead was observed in the first 500 mL of water after prolonged stagnation (Singh and Manivic, 1991). Gardels and Sorg (1989) demonstrated that 60–75% of total lead leached appeared in the first 125 mL of water after prolonged stagnation. For cold water the peak lead concentrations occurred in the first or second 25 mL sample and decreased exponentially with time thereafter. For hot water the peak lead concentration occurred in the second or third 25mL sample before decreasing exponentially (Gardels and Sorg, 1989). In a system where fully flushed water had a lead content of just 1.7 µg/L, removing just 125 mL of water from the tap every hour kept lead

concentrations elevated (35-52 µg/L) throughout the day (Gulson et al., 1997). Lytle and Schock (2000) showed a temporary exponential increase in lead concentration with stagnation time before the rate leveled off. After 10 hours of stagnation, approximately 50-70% of the maximum lead concentration had been achieved, although, lead levels continued to increase even after 90 hours of stagnation. Their results are shown in Figures 3-5 and 3-6. It should be noted that the shape of the stagnation-concentration curves was the same for all situations regardless of water quality.



Impact of stagnation time on lead and dissolved oxygen concentration in lead pipe (13 mm diameter) exposed to softened water in Study A.

Figure 3-5. The change in lead concentration vs. stagnation time. (Reprinted from Lytle and Schock, 2000).



Impact of stagnation time on lead and dissolved oxygen concentration in lead pipe (13 mm diameter) exposed to non-softened water in Study A.

Figure 3-6. The change in lead concentration vs. stagnation time. (Reprinted from Lytle and Schock, 2000).

Some concentrations of lead in drinking water are shown in Table 3-10. The lead standard for drinking water was set by the U.S. EPA in 1988 with a maximum allowable limit of 5 µg/L for water entering the distribution system (Frey, 1989).

Lead in drinking water can be either in particulate or soluble form. Lead can be in the form of aqueous ions or complexes, particularly when pH is low. Solids are the product of nonadherent corrosion deposits, eroded pieces of plumbing material, or background concentrations from the distribution system (Lytle et al., 1993). Lead particles are released when pH and alkalinity are low, and are typically in the form of hydrocerrusite scales (McNeill and Edwards, 2004). The lead products of corrosion are $CaCO_3$, $PbCO_3$, $Pb_3(CO_3)_2(OH_2$, $Pb_{10}(CO_3)_6(OH)_6O$, $Pb_5(PO_4)_3OH$, and PbO (Lytle et al., 1993; McNeill and Edwards, 2004).

Based on the conditions described above, models to predict drinking water lead concentrations have been proposed (e.g., Clement et al., 2000; Van Der Leer et al., 2002).

Table 3-10. Tap Water Concentrations of Lead

Water Conc. (µg/L)	Location	Residence Type	Description	Reference
20	Vancouver, Canada	Apartments	copper or plastic pipes	Singh and Manivic, 1991
13	Vancouver, Canada	Houses	copper or plastic pipes	Singh and Manivic, 1991
0.70	Arizona	Residences	-	Sofuoglu et al., 2003
0.32	Mexico/US border	Residences	-	Sofuoglu et al., 2003
16	Halifax, Canada	Houses	standing water	Moir et al., 1996
8	Halifax, Canada	Houses	running water	Moir et al., 1996
3	Halifax, Canada	Apartments	standing water	Moir et al., 1996
2	Halifax, Canada	Apartments	running water	Moir et al., 1996
6	Halifax, Canada	Public School	standing water	Moir et al., 1996
5	Halifax, Canada	Public School	running water	Moir et al., 1996
17	Dharan, Saudi Arabia	Community sites	PVC pipes	Sadiq et al., 1997
7.7	Clinton, MA	Residences	standing water	Isaac et al., 1997
25.0	Gardner, MA	Residences	standing water	Isaac et al., 1997
15.3	Fall River, MA	Residences	standing water	Isaac et al., 1997
11.6	New Bedford, MA	Residences	standing water	Isaac et al., 1997

3.4 EXPOSURE: FOOD INGESTION

Lead contaminated food is a major route of lead exposure. In one of the most thorough studies of lead ingestion in food, Flegal et al. (1990) showed that North Americans ingest an estimated 50 µg of lead each day through food, beverages, and dust, and approximately 30-50% of this amount is through food and beverages. The global average daily intake is about 80 µg/day from food and 40 µg/day from drinking water according to estimates made by the UN Environment Program (Juberg et al., 1997). In Australia, women between 20 and 39 years of age ingest between 7.3 and 9.7 µg/day (Gulson et al., 2001b). Infants that are breast-fed take in approximately 0.73 µg/day compared to 1.8 µg/day for formula-fed infants (Gulson et al., 2001b). Australian children ingest approximately 6.4 µg/day. A duplicate diet study shows that most diets contain a large amount of house dust (Manton et al., 2005). Other significant sources of lead in the diet are calcium-supplemented food where calcium is derived from limestone and lead in tin coatings. For U.S. children age 0-12 months, 13-24 months, 2-6 years, and their mothers, the estimated rate of lead ingestion was 1.8 µg/day, 3.3 µg/day, 4.1 µg/day and 7.5 µg/day respectively (Manton et al., 2005). This is significantly lower than the value reported above by Flegal et al. (1990), which may reflect the drop in lead emissions since the late 1980s.

The primary source of lead in food is atmospheric deposition (Flegal et al., 1990). Overall, anthropogenic aerosols account for an estimated 40% of lead in food, while the bulk of

the remainder is derived from harvesting, transport, processing, packaging, or preparation (Flegal et al., 1990; Juberg et al., 1997; Dudka and Miller, 1999). Lead contamination in poultry and livestock is also primarily atmospheric in origin. Lead deposits on forage or feed, or soil that is directly ingested (Flegal et al., 1990). Lead concentrations in food increase by a factor of 2 to 12 between harvest and consumption (Flegal et al., 1990). A food production facility in Turkey was shown to contaminate pasta with lead (Demirözü and Saldamli, 2002). Lead concentrations in the semolina were between 14.2 and 36.5 ng/g compared with the finished pasta product where concentrations ranged between 107.1 and 147.6 ng/g (Demirözü and Saldamli, 2002). A similar increase (from 0.5 ng/g to 230 ng/g) between raw and finished cocoa products has also been observed (Rankin et al., 2005). In this case, contamination seems to occur during shipping and/or processing.

Lead concentrations in vegetables may be increased by soil amendments such as mine wastes, slag, or fly ash. Historically, mine tailings were often disposed in streambeds, and this poses an exposure risk when sediments are harvested to boost productivity in gardens (Cobb et al., 2000). Slag is sometimes used for constructing agricultural and forestry roads or for landfill. This can be an additional source of lead contamination for nearby crops (Bunzl et al., 2001). Fly ash is applied to land infrequently for alkaline adjustment, as cover for landfills, or to amend agricultural soils. Elevated lead levels in fly ash can subsequently contaminate crops (Brake et al., 2004). Although soil contamination may be important on a local scale, atmospheric deposition is, overall, a more significant source of food lead than uptake from soil. For example, more than 52% of the total lead present in citrus fruits was removed by washing, indicating that surface deposits make up the bulk of lead contamination in unprocessed foods (Caselles, 1998).

The concentrations of lead measured in food are shown in Table 3-11. In general, food lead concentrations have decreased as a direct result of the decrease in airborne emissions of lead from automotive gasoline. This has been directly shown through measurements performed on vintage wines (Lobinski, 1995; Médina et al., 2000). The organolead concentration in French, Californian, Australian, and Argentinean wines peaked in 1978 (Lobinski, 1995). The maximum concentration was ~0.5 µg/L which was 10-100 times higher than lead concentrations in drinking water. Conversely, Médina et al. (2000) observed a peak lead concentration in French wine in

Table 3-11. The Concentration of Lead in Food Products

Food	Conc.	Location	Description	Reference
Barley, grain	0.4 ppm		Uncontaminated soil	Dudka and Miller (1999)
Barley, grain	2.0 ppm		Zn-Pb smelter contaminated	Dudka and Miller (1999)
Potato tubers, peeled	0.21 ppm		Uncontaminated soil	Dudka and Miller (1999)
Potato tubers, peeled	0.89 ppm		Zn-Pb smelter contaminated	Dudka and Miller (1999)
Lettuce	0.19 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Spinach	0.53 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Potatoes	0.03 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Wheat	0.02 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Rice	0.01 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Sweet corn	0.01 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Field corn	0.01 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Carrots	0.05 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Onions	0.04 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Tomatoes	0.03 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Peanuts	0.01 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Soybeans	0.04 ppm		Edible portion, untreated soil	Dudka and Miller (1999)
Applesauce, canned	8.5 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Fruit cocktail, canned	7.1 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Spinch, fresh	2.4 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Peaches, canned	6.0 µg/serving		FDA Total Diet Study	Juberg et al. (1997)

Table 3-11 (cont'd). The Concentration of Lead in Food Products

Food	Conc.	Location	Description	Reference
Pears, canned	4.9 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Strawberries, fresh	1.1 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Apple juice, bottled	2.6 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Wine	7.7 µg/serving		FDA Total Diet Study	Juberg et al. (1997)
Vaccinium vitis-idaea	0.4-2.3 ppm	Monchegorsk, Russia	Berry, near Ni-Cu smelter	Barcan et al. (1998)
Vaccinium myrtillus	0.7-1.6 ppm	Monchegorsk, Russia	Berry, near Ni-Cu smelter	Barcan et al. (1998)
Rubus chamaemorus	0.3-4.7 ppm	Monchegorsk, Russia	Berry, near Ni-Cu smelter	Barcan et al. (1998)
Empetrum hermaphroditum	0.3-1.5 ppm	Monchegorsk, Russia	Berry, near Ni-Cu smelter	Barcan et al. (1998)
Leccinum auranticcum	0.8-2.3 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Leccinum sacbrum	1.1-5.2 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Russul vesea	1.1-3.4 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Xerocomus subtomentosus	1.3-3.1 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Suillus luteus	2.0-2.3 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Lactarius trivialis	1.1-3.1 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Lactarius torminosus	0.6-3.5 ppm	Monchegorsk, Russia	Mushroom, near Ni-Cu smelter	Barcan et al. (1998)
Lettuce	0.65-1.3 ppm	Copenhagen, Denmark	Close to lead smelter	Moseholm et al. (1992))
Lettuce	0.15-0.46 ppm	Copenhagen, Denmark	Removed from lead smelter	Moseholm et al. (1992))
Lettuce	0.36 ppm	Copenhagen, Denmark	Background concentration	Moseholm et al. (1992))
Carrots	0.07-0.28 ppm	Copenhagen, Denmark	Close to lead smelter	Moseholm et al. (1992))
Carrots	<0.02-0.09 ppm	Copenhagen, Denmark	Removed from lead smelter	Moseholm et al. (1992))

Table 3-11 (cont'd). The Concentration of Lead in Food Products

Food	Conc.	Location	Description	Reference
Carrots	0.02-0.03 ppm	Copenhagen, Denmark	Background concentration	Moseholm et al. (1992))
Potatoes	<0.02-0.12 ppm	Copenhagen, Denmark	Close to lead smelter	Moseholm et al. (1992))
Potatoes	<0.02-0.06 ppm	Copenhagen, Denmark	Removed from lead smelter	Moseholm et al. (1992))
Potatoes	<0.02 ppm	Copenhagen, Denmark	Background concentration	Moseholm et al. (1992))
Kale	1.4-9.3 ppm	Copenhagen, Denmark	Close to lead smelter	Moseholm et al. (1992))
Kale	0.58-2.4 ppm	Copenhagen, Denmark	Removed from lead smelter	Moseholm et al. (1992))
Kale	0.52-0.72 ppm	Copenhagen, Denmark	Background concentration	Moseholm et al. (1992))
Wine	65 µg/L	France	Vintage 1990-1995	Médina et al. (2000)
Breast milk	0.55 µg/kg	Australia		Gulson et al. (2001a)
Infant formula	1.6 µg/kg	Australia		Gulson et al. (2001a)
Baby food	2.9 µg/kg	Australia		Gulson et al. (2001a)
Brassica juncea	298.3 ppm	Taihe, China	Indian mustard, near lead smelter	Cui et al. (2003)
Triticum aestivum L.	19.2 ppm	Taihe, China	Common wheat, near lead smelter	Cui et al. (2003)
Basil	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Cabbage	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Cilantro	49 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Collard greens	12 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Coriander	39 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Ipasote	14 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Lemon balm	20 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)

Table 3-11 (cont'd). The Concentration of Lead in Food Products

Food	Conc.	Location	Description	Reference
Mint	<10 - 60 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Mustard greens	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Parsley	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Red chard	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Rhubarb	<10 - 36 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Sage	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Swiss chard	22-24 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Thyme	<10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Carrot	10 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Onion	21 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Radish	12-18 ppm	Chicago, IL	Edible portion, urban garden	Finster et al. (2004)
Tuna, canned	0.1 ppm (max.)			Lourenço et al. (2004)
Sardines, canned	0.2 ppm (max.)			Lourenço et al. (2004)
Blue mussel, canned	0.3 ppm (max.)			Lourenço et al. (2004)
Balsamic vinegar	15-307 µg/L			Ndung'u et al. (2004)
Wine vinegar	36-50 µg/L			Ndung'u et al. (2004)
Tea leaves	0.59-4.49 ppm	Zhejiang Province, China	Commercial tea producing areas	Jin et al. (2005)
Cocoa beans	0.5 ng/g	Nigeria		Rankin et al. (2005)
Cocoa, manufactured	230 ng/g	Nigeria		Rankin et al. (2005)
Chocolate products	70 ng/g	Nigeria		Rankin et al. (2005)

the early 1950s. Isotopic analysis indicates that automotive emissions were the dominant source of lead contamination since 1950. It is not clear why French wine concentrations decreased through the late 1970s while automotive emissions were still increasing.

3.5 OTHER ROUTES OF EXPOSURE

3.5.1 Lead-Based Paint

Lead-based paint poses a potential exposure risk due to inhalation during renovation or demolition projects, or due to ingestion from hand-to-mouth activities and pica, which are common in children. As lead-based paint degrades, it becomes incorporated into house dust, which children ingest at a rate of ~100 mg per day (Flegal et al., 1990). Lead-based paint exposure is one of the most common causes of clinical lead toxicity. Lead-based paint can pose an inhalation risk during renovation and demolition activities. As described in Section 3.1 of this document, renovation projects often involve abrasive blasting techniques to remove old layers of paint. This breaks the lead into small pieces that are easily inhaled (Chute and Mostaghim, 1991; Mickelson and Johnston, 1995; Jacobs et al., 1998; Mielke et al., 2001). At industrial sites, exposure is limited primarily to workers. However, during residential renovation or abatement projects, residents may be unduly exposed to very high levels of airborne lead.

3.5.2 Calcium Supplements

Potentially toxic levels of lead were measured in calcium supplements in studies undertaken in the 1960s through the early 1990s (Scelfo and Flegal, 2000). An analysis of 136 different brands of supplements showed that two-thirds of the supplements did not meet the 1999 California criteria for acceptable lead levels: 1.5 µg/daily dose of calcium (Scelfo and Flegal, 2000). The lowest concentrations were observed in calcium products that were nonchelated synthesized and/or refined. These corresponded to antacids and infant formulas. Antacids and infant formulas had lead concentrations ranging from below the detection limit to 2.9 µg Pb/g calcium (Scelfo and Flegal, 2000). Natural calcium supplements derived from bonemeal, dolomite, or oyster shell were much more likely to be in exceedance of the 1999 standard. Lead levels reported elsewhere showed comparable lead levels in supplements and cow milk

(Juberg et al., 1997). Whole milk, 2% milk, and calcium supplements had lead concentrations in the range of 1.7-6.7 µg Pb/g calcium, 0.8-9.0 µg Pb/g calcium, and 3.1-6.9 µg Pb/g calcium, respectively (Juberg et al., 1997).

3.5.3 Glazes

Lead glazes have been commonly used throughout history. Kitchen glassware cannot have a lead solubility in excess of 2.5-7 µg/mL according to a 1980 rule by the Food and Drug Administration (Flegal et al., 1990). However, lead glazes on imported pottery may persist. Foods with low pH are particularly susceptible to solubilizing lead and contaminating food during storage in lead glazed glassware.

3.5.4 Miniblinds

Some imported vinyl miniblinds form lead dust upon disintegration (Juberg et al., 1997). This exposure route was responsible for several cases of lead poisoning in Arizona and North Carolina in the mid 1990s. Lead stabilizers are not used in vinyl miniblinds manufactured in the U.S. (Juberg et al., 1997).

3.5.5 Hair Dye

The analysis of Mielke et al. (1997) shows that some hair dyes contain lead acetate in the range of 2300-6000 µg Pb/g of product. This lead can be easily transferred via hand-to-mouth and hand-to-surface activity, and an estimated 3–5% of lead acetate can be transferred through the skin. Hair dyes tested in this study contain 3–10 times more lead than is allowable for paint (Mielke et al., 1997).

3.5.6 Other Potential Sources of Lead Exposure

Additional consumer products that may pose a risk of lead exposure include lead crystal, pool cue chalk (Miller et al., 1996), cosmetics, and folk remedies.

3.6 MEASUREMENT METHODS

Emissions estimates made through direct measurements can be done either using grab, periodic, or continuous monitoring. Determining the rate of emissions requires knowing both the fluid flow rate and the concentration of lead in the air (or water). Thus it is much easier to measure emissions from stacks than it is to measure fugitive, diffuse, or nonpoint emissions (Frey and Small, 2003).

The concentration of lead in air can be measured through several different methods. Use of filter media is one of the most common methods. Inertial impactors are another method, and in this case particles are separated by size. An additional method involves mounting a particle separation device in the stack along with gas flow control and metering equipment. Measurement of the mass collected in the cyclone hopper and the filter follows (Clarke and Bartle, 1998).

Sampling of airborne particles to determine quantity and species can be performed via direct-reading instruments, which include optical counters, electrical counters, resonant oscillation aerosol mass monitors, and beta radiation detection (Koutrakis and Sioutas, 1996). Additionally, particles may be collected in cyclones and denuder systems.

Collected particles are analyzed for lead using x-ray fluorescence analysis (XRF), proton-induced x-ray emission (PIXE), neutron activation analysis (NAA), or atomic absorption (AA) (Koutrakis and Sioutas, 1996).

Lead concentrations in soil, food, and other environmental media are determined using similar techniques. Generally substances undergo acid digestion in an HCl or HNO3 solution before analysis via XRF, PIXE, NAA, or AA.

3.7 SUMMARY

Concentrations of lead in all environmental media are highly elevated in urban areas. A comprehensive analysis of multimedia concentrations of lead showed that people in cities, especially in poor and minority-dominated neighborhoods, are the most at risk for lead exposure (Chadha et al., 1998).

The highest air, soil, and road dust concentrations are found near major lead sources such as smelters, mines, and heavily trafficked roadways. Airborne concentrations have declined

dramatically with the phase out of leaded gasoline. Soil concentrations have remained relatively constant.

Drinking water is susceptible to lead contamination primarily through leaching from pipes, solder, and faucets. Water that has been stagnant in pipes, been disinfected with chloramines, has a low pH, or has a low alkalinity are particularly high risk for leaching lead into drinking water.

Lead-contaminated food is a major exposure route. Deposition of airborne lead and house dust are the major sources of lead in food. Significant quantities of lead are ingested by certain populations every day.

Other sources of lead exposure vary in their prevalence and potential risk. These include lead-based paint, calcium supplements, lead-based glazes, some kinds of miniblinds, hair dye, and other consumer products.

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4. MODELS OF HUMAN EXPOSURE THAT PREDICT TISSUE DISTRIBUTION OF LEAD

4.1 OBJECTIVES IN MODELING LEAD EXPOSURE AND TISSUE DISTRIBUTION OF LEAD

Models are essential for quantifying human health risks that derive from exposures to lead. Dose-response relationships for nearly all of the major health effects of lead in humans are expressed in terms of internal dose (e.g., blood or bone lead concentrations). Application of this internal dose-response information to the assessment of risks from environmental exposures to lead requires a way of relating internal dose to levels of lead in the environmental media to which humans come in contact (e.g., air, water, surface dust). Models provide the only means for accomplishing this objective. Models come in various forms. Multivariate regression models, commonly used in epidemiology, provide estimates of the contribution of variance in the internal dose metric to various determinants or control variables (e.g., surface dust lead concentration, air lead concentration). Structural equation modeling links several regression models together to estimate the influence of determinants on the internal dose metric. Regression models can provide estimates of the rate of change of blood or bone lead concentration in response to an incremental change in exposure level (i.e., slope factor). The strength of regression models is that they can have relatively few parameters, which allows a rigorous quantitative assessment of uncertainty in the slope factor. However, the simplicity of regression models also frequently excludes numerous parameters that are known to influence human lead exposures and the relationship between human exposure and tissue lead levels, parameters which are expected to vary spatially and temporally. Thus, extrapolation of regression models to other spatial or temporal contexts, which is often necessary for regulatory applications of the models, can be problematic.

An alternative to regression models are mechanistic models, which attempt to specify all parameters needed to describe the mechanisms (or processes) of transfer of lead from the environment to human tissues. Such mechanistic models are more complex than regression models, which introduces challenges in terms of their mathematical solution. However, by incorporating parameters that can be expected to vary spatially or temporally, or across

individuals or populations, mechanistic models can be extrapolated to a wide range of exposure scenarios. Exposure-biokinetic models, a type of mechanistic models, are highly simplified mathematical representations of relationships between levels of lead in environmental media and human lead intakes (e.g., μg lead ingested per day). These models include parameters representing processes of lead transfer between environmental media (e.g., air to surface dust) and to humans, including rates of human contact with the media and intakes of the media (e.g., g soil ingested per day). Biokinetic models provide the analogous mathematical representation of relationships between lead intakes and levels of lead in body tissues (e.g., blood lead concentration); and they include parameters that represent processes of lead transfer (a) from portals of entry into the body and (b) from blood to tissues and excreta. Exposure-biokinetic models provide an approach for predicting blood lead concentrations (or lead concentrations in other tissues) that corresponds to a specified exposure (medium, level, and duration). Detailed information on exposure and internal dose can be obtained from controlled experiments, but almost never from epidemiological observations or from public health monitoring programs. Exposure-biokinetic models can provide these predictions in the absence of complete information on the exposure history and blood lead concentrations for an individual (or population) of interest. Therefore, these models are critical to applying epidemiologically-based information on blood lead-response relationships to the quantification and characterization of human health risk. They are also critical for assessing the potential impacts of public health programs directed at mitigation of lead exposure or of remediation of contaminated sites.

Mechanistic models also have several other important features that are useful for risk assessment and for improving our basic understanding of lead exposures and biokinetics. They integrate complex information on lead exposure and biokinetics into a form that provides predictions, rather than just an organized grouping of observations. By analyzing the relationships between model assumptions and predictions (i.e., sensitivity analysis), and by comparing predictions to observations (i.e., model evaluation), such models can contribute to the identification of important gaps in our understanding of lead exposure, biokinetics, and risk. Thus, these models provide a consistent method for making, evaluating and improving predictions that support risk assessment and risk management decisions.

Modeling of human lead exposures and biokinetics has advanced considerably during the past several decades. Among the most important new advances are development, evaluation, and

extensive application of the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children (U.S. Environmental Protection Agency, 1994a) and the development of models that simulate lead biokinetics in humans from birth through adulthood (Leggett, 1993; O'Flaherty 1993, 1995). While these developments represent important conceptual advances, several challenges remain for further advancements in modeling and applications to risk assessment. The greatest challenge derives from the complexity of the models. Human exposure-biokinetics models include large numbers of parameters, which are required to describe the many processes that contribute to lead intake, absorption, distribution, and excretion. The large number of parameters complicates the assessment of confidence in parameter values, many of which cannot be directly measured. Statistical procedures can be used to evaluate the degree to which model outputs conform to "real-world" observations and values of influential parameters can be statistically estimated to achieve good agreement with observations. Still, large uncertainty can be expected to remain about many, or even most, parameters in complex exposure-biokinetic models such as those described below.

Given the difficulty in quantitatively assessing uncertainty in values of all of the individual parameters in an exposure-biokinetics model, assurance that the model accurately represents the real-world in all aspects is virtually impossible. As consequence of this, Oreskes (1998) noted, *"…the goals of scientists working in a regulatory context should be not validation but evaluation, and where necessary, modification and even rejection. Evaluation implies an assessment in which both positive and negative results are possible, and where the grounds on which a model is declared, good enough are clearly articulated."* In this context, evaluation of confidence in a given exposure-biokinetic model rests largely on assessment of the degree to which model predictions, based on model inputs appropriate for a situation, conform to observations and/or expectations; and, most importantly, the degree to which this conformity does or does not satisfy requirements of model application to a specific context. Because of limitations in observations of predicted outcomes, it may be possible to evaluate confidence in some uses of a model, but not others. Similarly, it is possible for confidence in a model to be judged acceptable for a given use, but not for others. The concept of *validation* of highly complex mechanistic models, outside of the context of a specific use of the model, has little meaning.

In the ensuing discussion of specific models, reported efforts to evaluate the models are noted. In most cases, however, the relevance of these evaluations to the assessment of confidence in a specific use of that model (e.g., predicting average blood lead concentrations in children who live in areas that have certain cross-sectionally measured environmental lead levels) cannot be ascertained from the reported literature. Nevertheless, as a framework for qualitatively comparing the various evaluative procedures that have been applied, the following general classification of model evaluations has been adopted:

- Sensitivity analysis has been conducted and most influential parameters identified and uncertainty characterized.
- Model predictions have been compared qualitatively to observations.
- Predictions have been compared quantitatively to observations (i.e., a statistical model has been applied for estimation of "goodness of fit" and uncertainty).
- Confidence in model predictions for specific uses has been quantitatively evaluated.
- Accuracy of model implementation code has been verified.

In the sections that follow, an overview is provided with regard to the evolution of important lead biokinetic modeling aspects that constitute major modeling advances during the past 25 years or so leading to the development of EPA's All Ages Lead Model (AALM) discussed below in Section 4.6. Descriptions of the individual models are intended to provide only brief snapshots of key features of each model, with particular attention to conceptual features that are unique to each model. Key references are cited in which more complete specifications of model parameters can be found.

4.2 HISTORIC OVERVIEW OF LEAD MODELS

4.2.1 Rabinowitz Model

Early lead modeling applications presented lead biokinetics in classical pharmacokinetics terms. Compartments represented kinetically homogeneous pools of lead which might be associated with individual organs or groups of organs. Among the first of such models was one proposed by Rabinowitz et al. (1976) based on a study of the kinetics of ingested stable lead isotope tracers and lead mass balance data in five healthy adult males (Figure 4-1). The

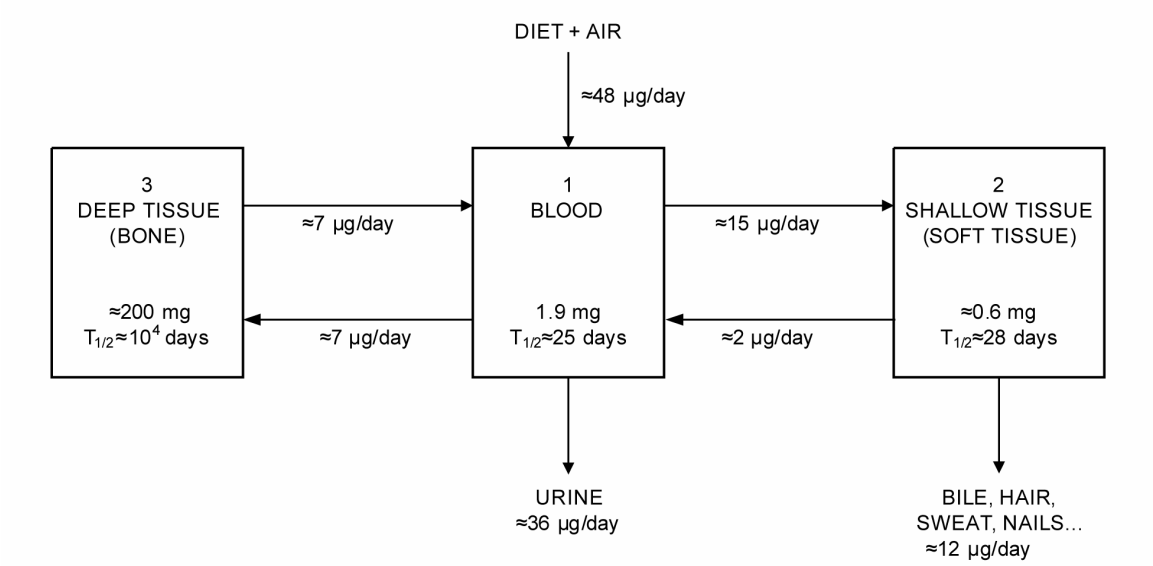


Figure 4-1. Lead biokinetics based on Rabinowitz et al. (1976). Half-times are based on reported residence times for compartments 1, 2, and 3: 25, 28, and 10^4 days, respectively (half-time ln(2)/(1/residence time).

Rabinowitz model has three compartments: (1) a central compartment representing blood and other tissues and spaces in rapid equilibrium with blood (e.g., interstitial fluid); (2) a shallow tissue compartment, representing soft tissues and rapidly exchanging pools within the skeleton; and (3) a deep tissue compartment, representing, primarily, slowly exchanging pools of lead within bone. Excretion pathways include urinary (from the central compartment) and bile, sweat, hair, and nails (from the shallow tissue compartment). The model predicts pseudo-first order half-times for lead of approximately 25, 28, and 10^4 days in the central, shallow tissue, and deep compartments, respectively (these values were calculated based on reported residence times, the reciprocal of the sum of the individual elimination rate constants). The slow kinetics of the deep tissue compartment leads to the prediction that it would contain most of the lead burden following chronic exposures (e.g., for years), consistent with lead measurements made in human autopsy samples (Barry, 1975; Gross et al., 1975; Schroeder and Tipton, 1968). Note that this model did not simulate the distribution of lead within blood (e.g., erythrocytes and plasma), nor did it simulate subcompartments within bone or physiological processes of bone turnover that might affect kinetics in the deep tissue compartment.

4.2.2 Marcus Model(s)

Marcus (1985b) reanalyzed the data from stable isotope tracer studies of Rabinowitz et al. (1976) and derived an expanded multicompartment kinetic model for lead (Figure 4-2). The model included separate compartments with different lead turnover rates for cortical (slow, $t_{1/2} = 1.2 \times 10^4$ to 3.5×10^4 days) and trabecular (fast, $t_{1/2} = 100$ to 700 days) bone, an approach subsequently adopted in several other models (O'Flaherty, 1995; U.S. Environmental Protection Agency, 1994a,b; Leggett, 1993; O'Flaherty, 1993; Bert et al., 1989). A more complex representation of the lead disposition in bone included explicit simulation of lead diffusion within the bone volume of the osteon and exchange with blood at the canaliculus (Marcus, 1985a; Figure 4-3). Lead diffusion in bone was based on lead kinetics data from studies conducted in dogs. A similar approach to simulating radial diffusion of lead in bone, expanded to include eight concentric diffusion shells, was implemented by O'Flaherty (1995, 1993). Marcus (1985c) also introduced nonlinear kinetics of exchange of lead between plasma and erythrocytes. The blood kinetics included four blood subcompartments: diffusible lead in plasma, protein-bound lead in plasma, a "shallow" erythrocyte pool, and a "deep" erythrocyte pool (see Figure 4-4). The Marcus (1985c) model predicted the curvilinear relationship between plasma and blood lead concentrations that has been observed in humans (DeSilva, 1981).

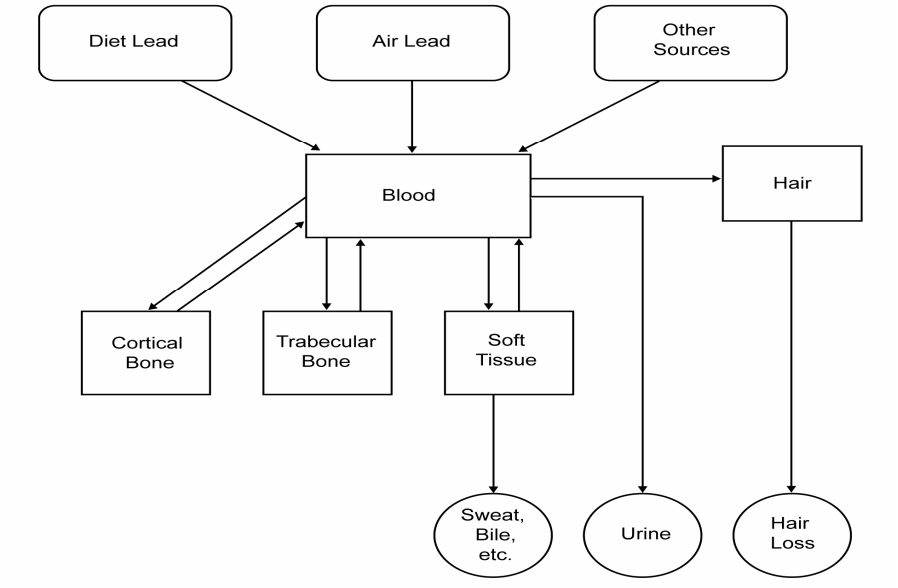


Figure 4-2. Lead biokinectics based on Marcus (1985b). Bone is represented as a slow turnover (cortical) compartment and a faster (trabecular) compartment.

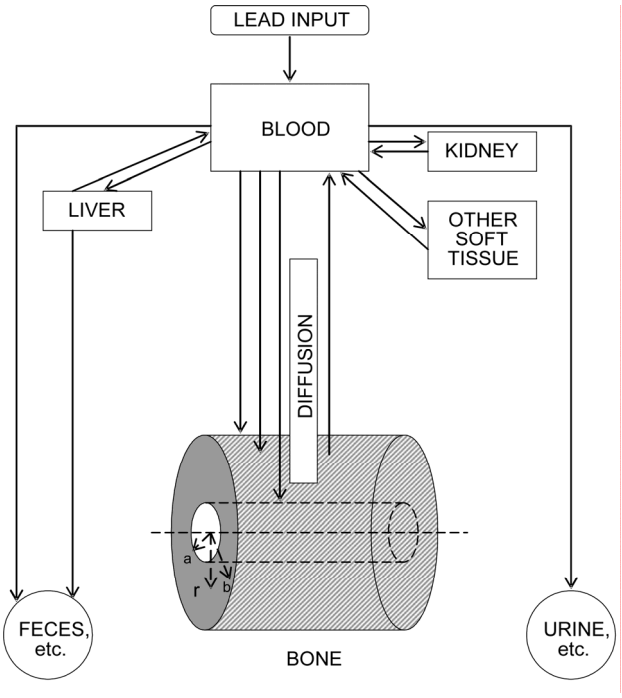


Figure 4-3. Lead biokinetics based on Marcus (1985a). Bone is represented as an extended cylindrical canalicular territory. The canalicular territory has a radius b, and surrounds the canaliculus of radius a. Lead diffuses across radius library, between the fluid in the canaliculus (which is in communication with blood in the Haversian canal, not shown) and the bone volume of the canalicular territory.

4.2.3 Bert Model

Bert et al. (1989) adopted the bone model from Marcus (1985b), in which the bone compartment is subdivided into slow cortical bone and faster trabecular bone compartments (Figure 4-5). The central compartment (denoted as *blood*) is assumed to be 1.5 times the volume of whole blood, with the whole blood volume varying in direct proportion with body weight. The model includes a discrete pathway for excretion of unabsorbed lead from the gastrointestinal (GI) tract into feces. Secretion of lead in bile, gastric secretions, and saliva are represented as transfers from the soft tissue compartment to the GI tract. Compartment transfer coefficients were based on average values estimated for four individuals from the Rabinowitz et al. (1976) study. Initial average values for lead in cortical bone for a given age at the start of a simulation were derived from Barry (1975).

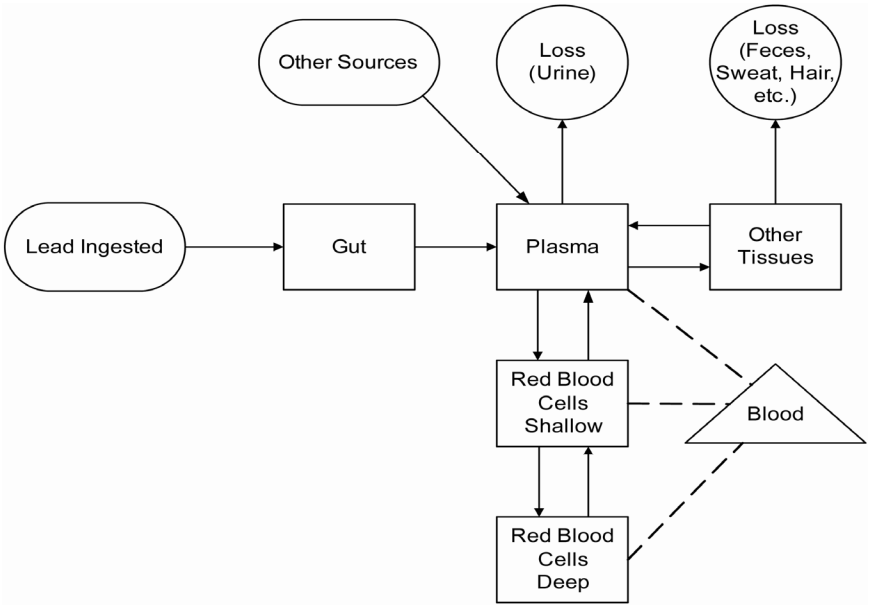


Figure 4-4. Lead biokinetics based on Marcus (1985c). Blood is represented with a plasma (central exchange) compartment and a red blood cell compartment, the latter having shallow and deep pools.

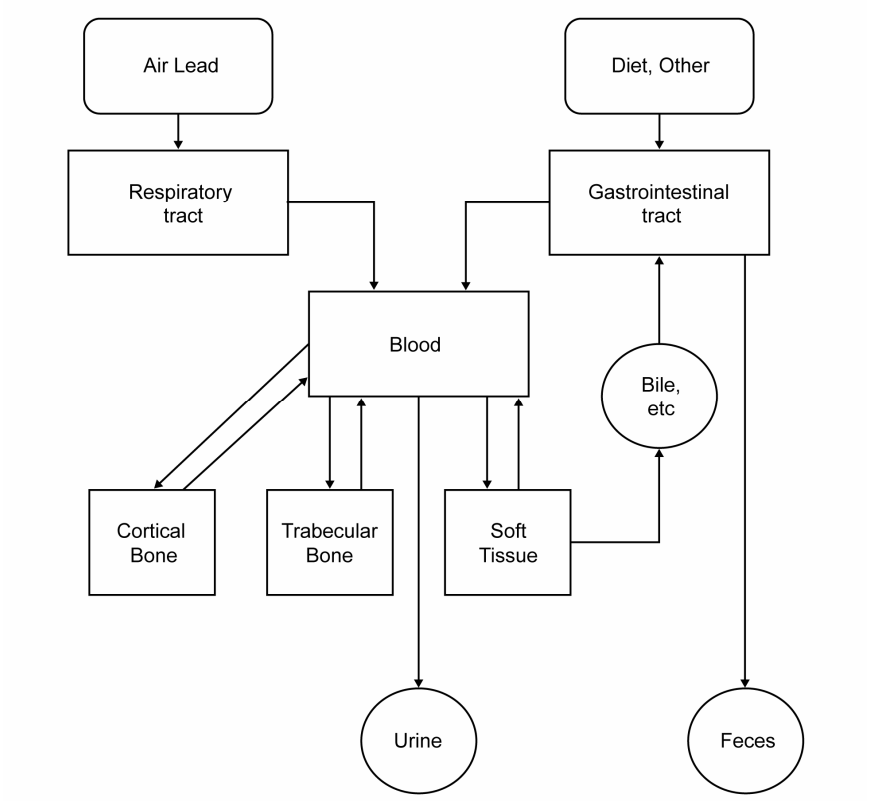


Figure 4-5. Lead biokinetics based on Bert et al. (1989).

4.2.4 Contemporary Models

Additional information on lead biokinetics, bone mineral metabolism, and lead exposures has led to further refinements and expansions of these earlier modeling efforts. In particular, three pharmacokinetic models are currently being used or are being considered for broad application in lead risk assessment: (1) the Integrated Exposure Uptake BioKinetic (IEUBK) model for lead in children developed by EPA (U.S. Environmental Protection Agency, 1994a,b; White et al., 1998); (2) the Leggett model, which simulates lead kinetics from birth through adulthood (Leggett, 1993); and (3) the O'Flaherty model, which simulates lead kinetics from birth through adulthood (O'Flaherty, 1995, 1993). Of the three approaches, the O'Flaherty model has the fewest lead-specific parameters and relies more extensively on physiologically based parameters to describe volumes, flows, composition, and metabolic activity of blood and bone that determine the disposition of lead in the human body. Both the IEUBK model and the Leggett model are more classical multicompartmental models; that is, the values for the age-specific transfer rate constants for lead are based on kinetics data obtained from studies conducted in animals and humans and may not have precise physiological correlates. Thus, the structure and parameterization of the O'Flaherty model is distinct from both the IEUBK model and Leggett model. All three models represent the rate of uptake of lead (i.e., amount of lead absorbed per unit of time) as relatively simple functions (f) of lead intake:

$$Uptake = Intake \cdot AF \quad (4\text{-}1)$$

$$Uptake = Intake \cdot f_{(Intake)} \quad (4\text{-}2)$$

Values assigned to absorption factor (AF) or other variables in $f_{(Intake)}$ are, in general, age-specific and environmental medium-specific in some models. However, the models do not modify the representation of uptake as functions of the many other physiologic variables that may affect lead absorption (e.g., nutritional status). While one can view this approach as a limitation of the models, it also represents a limitation of the data available to support more complex representations of lead absorption.

The IEUBK model simulates multimedia exposures, uptake, and kinetics of lead in children ages 0–7 years; the model is not intended for use in predicting lead pharmacokinetics in adults. The O'Flaherty and Leggett models are lifetime models, and include parameters that simulate uptake and kinetics of lead during infancy, childhood, adolescence, and adulthood. Lead exposure (e.g., residence-specific environmental lead concentrations, childhood activity patterns) is not readily described by current versions of the O'Flaherty and Leggett models. By contrast, the IEUBK model includes parameters for simulating exposures and uptake to estimate average daily uptake of lead (μg/day) among populations of children potentially exposed via soil and dust ingestion, air inhalation, lead-based paint chip ingestion, tap water ingestion, and/or diet.

The above three models have been individually evaluated, to varying degrees, against empirical physiological data on animals and humans and data on blood lead concentrations in individuals and/or populations (U.S. Environmental Protection Agency, 1994a,b; Leggett, 1993; O'Flaherty, 1993). However, applications in risk assessment typically require that the models accurately predict blood lead distributions in real populations, in particular those values or percentages falling in the "upper tails" (e.g., ≥95th percentiles of the distributions, when input to the models consists of data that describe site-specific exposure conditions (e.g., environmental lead concentrations, physicochemical properties of soil and dust) (Beck et al., 2001; Griffin et al., 1999a,b). In evaluating models for use in risk assessment, exposure data collected at hazardous waste sites have been used to drive model simulations (Bowers and Mattuck, 2001; Hogan et al., 1998). The exposure module in the IEUBK model makes this type of evaluation feasible.

4.3 INTEGRATED EXPOSURE UPTAKE BIOKINETIC (IEUBK) MODEL FOR LEAD IN CHILDREN

4.3.1 Model Structure

The IEUBK model for lead in children (see Figure 4-6) is a multicompartmental pharmacokinetics model linked to an exposure and probabilistic model of blood lead concentration distributions in children (U.S. Environmental Protection Agency, 1994a,b; White et al., 1998). The model simulates exposure and biokinetics of lead from birth to age 7 years

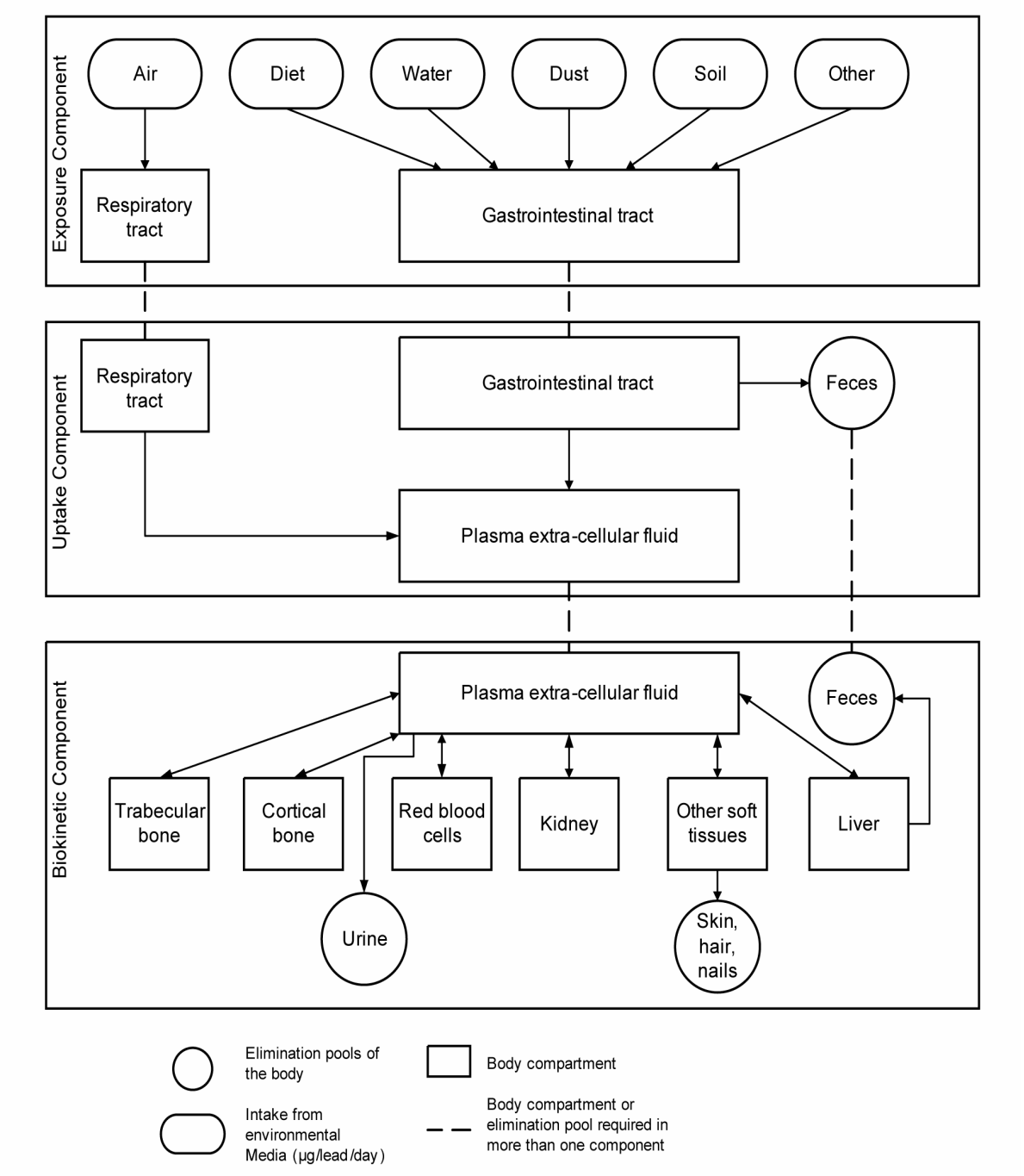


Figure 4-6. Structure of the integrated exposure uptake biokinetics model for lead in children (U.S. Environmental Protection Agency, 1994a,b; White et al., 1998).

(84 months) and was developed for predicting average quasi-steady state blood lead concentrations corresponding to daily average exposures, averaged over periods ≥1 year.

The model has four major components or submodels:

- Exposure model, in which average daily intakes of lead (μg/day) are calculated for each inputted exposure concentration (or rates) of lead in air, diet, dust, soil, and water;
- Uptake model, which converts environmental media-specific lead intake rates calculated from the exposure model into a media-specific time-averaged rates of uptake (μg/day) of lead to the central compartment (blood plasma);
- Biokinetic model, which simulates the transfer of absorbed lead between blood and other body tissues, elimination of lead from the body (via urine, feces, skin, hair, and nails), and predicts an average blood lead concentration for the exposure time period of interest; and
- Blood lead probability model, which simply applies a log-normal distribution (with specific geometric mean and geometric standard deviation parameters) to predict probabilities for the occurrence of a specified blood lead concentration in a population of similarly exposed children.

Exposure Model. The exposure model simulates intake of lead (μg/day) for exposures to lead in air (μg/m^3), drinking water (μg/L), soil-derived dust (μg/g), and diet (μg/day). The temporal resolution of the exposure model is 1 year; exposure inputs are intended to represent annual averages for an age-year time step (e.g., ages 1, 2, 3...years). Exposure inputs that represent the average daily value for an age-year will yield corresponding daily average intakes for the same age-year. The spatial resolution of the exposure model was intended to be a child's residence (e.g., the home and yard). The model accepts inputs for media intake rates (e.g., air volume breathing rates, drinking water consumption rate, soil and dust ingestion rate). The air exposure pathway partitions exposure to outdoor air and indoor air; with age-dependent values for time spent outdoors and indoors (hours/day). Exposure to lead in soil derived dust is also partitioned into outdoor and indoor contributions. The intakes from all ingested exposure media (diet, drinking water, soil-derived dust) are summed to calculate a total intake to the gastrointestinal tract, for estimating capacity-limited absorption (see description of the Uptake Model).

Uptake Model. The uptake model simulates lead absorption in the gastrointestinal tract as the sum of a capacity-limited (represented by a Michaelis-Menten type relationship) and

unlimited processes (represented by a first-order, linear relationship). These two terms are intended to represent two different mechanisms of lead absorption, an approach that is in accord with limited available data in humans and animals that suggest a capacity limitation for lead absorption (Mushak, 1991). One of the parameters for the capacity-limited absorption process (that represents that maximum rate of absorption) is age-dependent. The above representation gives rise to a decrease in the fractional absorption of ingested lead as a function of total lead intake as well as age. Absorption fractions are also medium specific (Figure 4-7). At 30 months of age, at low intakes (<200 μg/day), below the rates at which capacity-limitation has a significant impact on absorption, the fraction of ingested lead in food or drinking water that is absorbed is 0.5 and decreases to approximately 0.11 at high intake (>5000 μg/day). For lead ingested in soil or dust, fractional absorption is 0.35 at low intake (<200 μg/day) and decreases to 0.09 at high intake (>5000 μg/day).

The uptake model assumes that 32% of inhaled lead is deposited in the respiratory tract. This value was originally assigned based on a scenario of exposure to active smelter emissions, which assumed the particle size distribution in the vicinity of an active lead smelter; size-specific deposition fractions for the nasopharyngeal, tracheobronchial, and alveolar regions of the respiratory tract; and region-specific absorption fractions (Table 4-1). There are three compartments in the Leggett model for lung absorption, and these are distinguished by the depth to which the air particle penetrates. Lead deposited in the alveolar region is assumed to be completely absorbed from the respiratory tract, whereas, lead deposited in the nasopharyngeal and tracheobronchial regions is assumed to be transported to the gastrointestinal tract where absorption (approximately 30%) occurs.

Biokinetics Model. The biokinetics model includes a central compartment, plasma and extracellular fluid combined (plasma-ECF), six peripheral body compartments, and three elimination pathways. The temporal resolution of the biokinetics model is 1 month and, as discussed below, parameter values for bone-plasma-ECF exchanges were assigned with the objective of simulating the quasi-steady state condition of months, rather than short-term kinetics of days. The body compartments include kidney, liver, trabecular bone, cortical bone, and other soft tissue. The model simulates growth of the body and tissues, compartment volumes, and lead masses and concentrations in each compartment. Blood lead concentration at birth (neonatal) is assumed to be 0.85 of the maternal blood lead. Neonatal lead masses and concentrations are

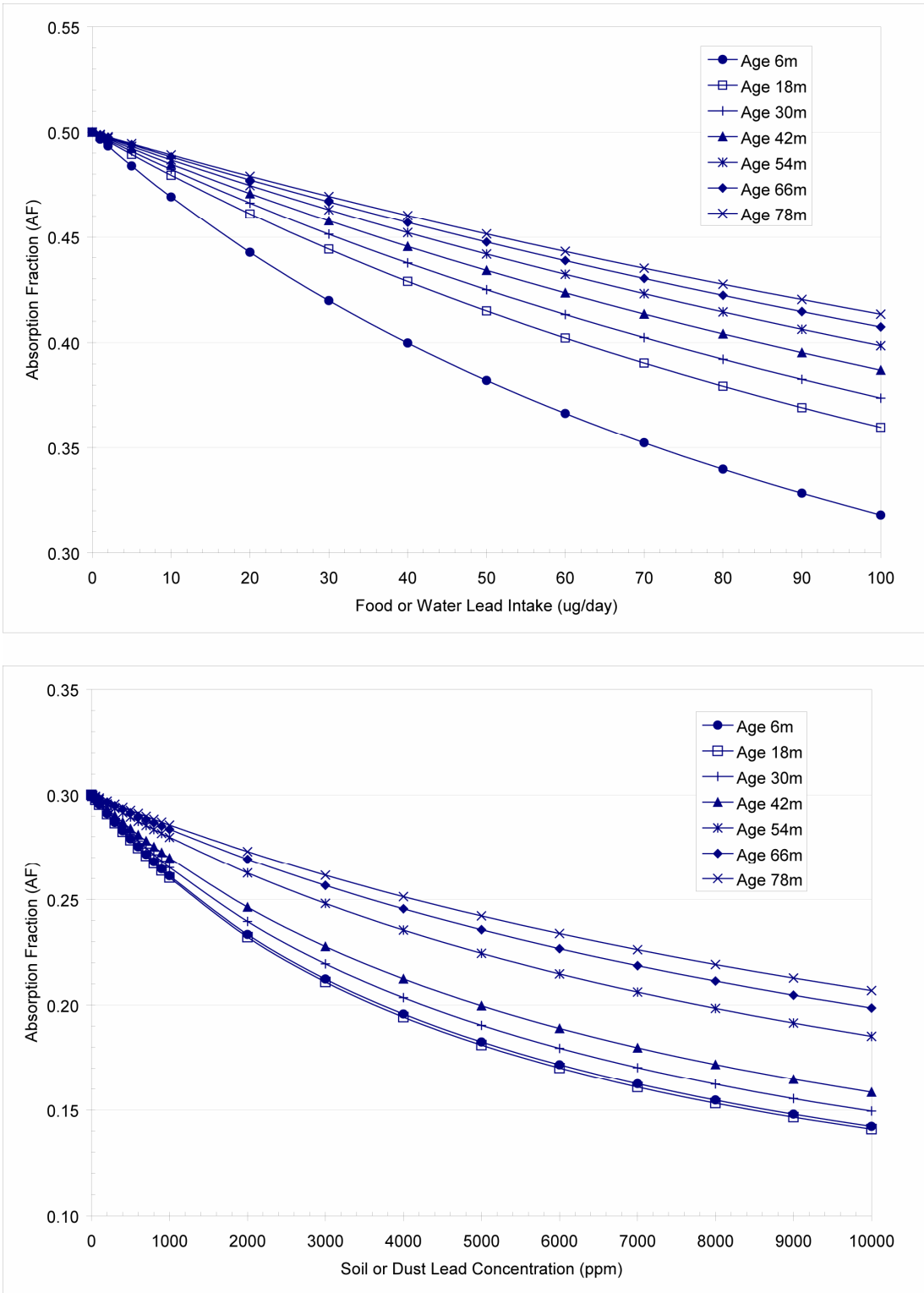


Figure 4-7. Age-dependency of absorption fraction for ingested lead in the IEUBK model for lead in children. Absorption fraction for food and water (top panel); soil and dust (bottom panel).

Table 4-1. Comparison of Slope Factors in Selected Slope Factor Models

Model	Receptor	Slope Factor		Absorption Fraction
		Intake	Uptake	
Bowers et al. (1994)	Adult	N/A	0.375	0.08
Carlisle and Wade (1992)	Child	Soil/dust: 0.07 Water: 0.04	N/D	N/D
Carisle and Wade (1992)	Adult	Soil/dust: 0.018 Water: 0.04	N/D	N/D
U.S. Environmental Protection Agency (1996)	Adult	ND	0.4	0.12
Stern (1994)	Child	Residential: T(0.056, 0.16, 0.18)	N/D	N/D
Stern (1994)	Adult	Nonresidential: U(0.014, 0.034)	N/D	

N/D = Not determined; T = triangular probability distribution function (PDF); U = uniform PDF

assigned to other compartments based on a weighted distribution of the neonatal blood lead concentration. Exchanges between the central compartment and tissue compartments are simulated as first-order processes, which are parameterized with unidirectional, first-order rate coefficients. Rate coefficients are allometrically scaled as a power function of body weight ($BW^{0.33}$).

Saturable uptake of lead into erythrocytes is simulated, with a maximum erythrocyte lead concentration of 120 μg/L. Excretory routes simulated include urine, from the central compartment; bile-feces, from the liver; and a lumped excretory pathway representing losses to skin, hair and nails, from the "other soft tissue" compartment.

Bone is simulated as a trabecular bone compartment (20% of bone volume) and a cortical bone compartment (80% of bone volume). Rate constants for transfer from plasma to the two bone compartments are assigned values that result in a 4:1 cortical lead:trabceular lead mass ratio within one biokinetic time step (one month). This is achieved by assigning the two bone compartments identical rate coefficients for transfer of lead from bone to plasma-ECF (half-time 8.5 days, at age 2 years), and *faster* (cortical, half-time 0.0083 days) and *slower* transfer (trabecular, half-time 0.035 days) from the plasma-ECF (cortical:trabecular rate ratio is

approximately 4:1). Note, this approach is different from previous and subsequent modeling approaches, in which cortical bone-to-plasma (or blood) transfer is assumed to occur slowly, relative to trabecular bone-to-plasma transfer (Marcus, 1985b; Bert et al., 1989; Leggett, 1993; O'Flaherty, 1993, 1995). For predictions of quasi-steady state conditions and the intended use of the IEUBK Model, the two general approaches can be expected to yield similar distributions of lead between the cortical and trabecular bone compartments.

Blood Lead Probability Model. Inputs to the IEUBK model are exposure point estimates that are intended to represent time-averaged central tendency exposures. The output of the model is a central tendency estimate of blood lead concentration for children who might experience the inputted average exposures. However, within a group of similarly exposed children, blood lead concentrations would be expected to vary among children as a result of inter-individual variability in media intakes (e.g., daily average intakes of soil-derived dust, drinking water, or food), absorption, and biokinetics. The model simulates the combined impact of these sources of variability as a lognormal distribution of blood lead concentration for which the geometric mean (GM) is given by the central tendency blood lead concentration outputted from the biokinetics model, and the geometric standard deviation (GSD) is an input parameter. The resulting lognormal distribution also provides the basis for predicting the probability of occurrence of given blood lead concentration within a population of similarly exposed children:

P_X = probability of exceeding a blood lead concentration of X µg/dL

P_{10} = probability of exceeding a blood lead concentration of 10 µg/dL

The model can be iterated for varying exposure concentrations (e.g., a series of increasing soil lead concentration) to predict the media concentration that would be associated with a probability of 0.05 for the occurrence of a blood lead concentration exceeding 10 µg/dL ($P_{10} = 0.05$).

4.3.2 Model Calibration and Evaluation

An evaluation of the IEUBK model has been carried out by comparison of model predictions of blood lead concentrations in children with observations from epidemiologic studies of hazardous waste sites (Hogan et al., 1998). Data characterizing residential lead

exposures and blood lead concentrations in children living at four Superfund National Priorities List (NPL) sites were collected in a study designed by the Agency for Toxic Substances and Disease Registry (ATSDR) and EPA. The residential exposure data were used as inputs to the IEUBK model and predicted blood lead concentration distributions were compared to the observed distributions in children living at the same residences. The IEUBK model predictions of geometric mean blood lead concentrations for children whose exposures were predominantly from their residence (i.e., no more than 10 hours/week away from home) were within 0.7 µg/dL of the observed geometric mean at each site. The prediction of the percentage of children expected to have blood lead concentrations exceeding 10 µg/dL were within 4% of the observed percentage at each site. This evaluation supports IEUBK model use for estimating blood lead concentrations in children at sites where their residential exposures can be adequately characterized. Similar empirical comparisons have shown that agreement between IEUBK model predictions and observed blood lead concentrations at specific locations is influenced by numerous factors, including (a) the extent to which the exposure and blood lead measurements are adequately matched and (b) site-specific factors (e.g., soil characteristics, behavior patterns, bioavailability) that may affect lead intake or uptake in children (Bowers and Mattuck, 2001; TerraGraphics Environmental Engineering, Inc., 2001).

4.3.3 Model Applications

Biomarkers Simulated. The IEUBK model computes masses of lead in bone and various soft tissues, and excretion of lead, which are used in the computation of blood lead concentration. However, the model was not developed for the purpose of predicting lead masses in these tissues or excreta. Blood lead concentration is the only lead biomarker output that is accessible to the user.

Exposure Inputs. The IEUBK model was developed to predict the probability of elevated blood lead concentrations in children exposed to user-specified annual average exposures to lead in air, food, drinking water, soil, and dust. As noted above, the exposure model has an age-year time step (the smallest time interval for a single exposure event) and, therefore, is more suited to applications in which long-term (i.e., ≥1year) average exposures and quasi-steady state blood lead concentrations are to be simulated. Intermittent exposures occur for brief periods of time (e.g., a weekend at the beach), or in cases where significant seasonal variations are different from

the typical residential or occupational exposure. In these cases, the IEUBK can accept time-weighted average exposures using the guidance provided in Syracuse Research Corporation (SRC) (2003). Shorter-term dynamics of blood lead concentration, that may result from exposures that are highly variable on time scales of days or weeks, will not be captured with this approach (Lorenzana et al., 2005; Khoury and Diamond, 2003).

Modeling Variability and Uncertainty. As noted above, the IEUBK model uses a lognormal probability model to simulate inter-individual variability in blood lead concentrations attributable to variability in media intakes, absorption, and biokinetics. The model uses a generic default value of 1.6 for the blood lead concentration individual GSD (GSD_i). This value was derived from an analysis of exposure (soil lead)-stratified variability in blood lead concentrations in various cohorts of children (U.S. Environmental Protection Agency, 1994a; White et al., 1998). Griffin et al. (1999b) also explores various statistical methods for estimating for estimating an appropriate GSD_i (regression, box modeling, structural equation modeling).

A Monte Carlo approach has been used to simulate and propagate variability and uncertainty in exposure and absorption through IEUBK model simulation of blood lead concentrations (Goodrum et al., 1996). This extension of the model provides an alternative to the generic blood lead probability approach for incorporating explicit estimates of variability (and uncertainty in variability) in exposure and absorption into predictions of an expected probability distribution of blood lead concentrations. A quantitative uncertainty analysis of IEUBK model-based estimates of the P_{10} for a smelter site in Utah revealed that parameters specifying soil ingestion rate were a dominant contributor to uncertainty in the P_{10}; however, the contribution of soil ingestion uncertainty, relative to uncertainty in other model parameters (i.e., mean soil lead concentration, absorption fraction) varied across individual locations (Initial Study Zones) at the site (Griffin et al., 1999a).

4.3.4 Validation/Verification of IEUBK

The IEUBK model was initially released to the public in 1994 as a compiled DOS-based C program (IEUBK v99d). This version was subjected to an independent validation and verification study which verified that the code accurately implement the model (Mickle, 1998; Zaragoza and Hogan, 1998). A 32-bit C++ (IEUBKwin32) version of the model is available for download from an EPA website (http://www.epa.gov/superfund/programs/lead/ieubk.htm).

4.4 LEGGETT MODEL

4.4.1 Model Structure

The Leggett model was developed from a biokinetic model originally developed for the International Commission on Radiological Protection (ICRP), for calculating radiation doses from environmentally important *bone-seeking* radionuclides, including radioisotopes of lead (Leggett, 1985, 1992a,b). The model has been used to develop cancer risk coefficients for internal radiation exposures to lead and other alkaline earth elements that have biokinetics similar to those of calcium (ICRP, 1993; U.S. Environmental Protection Agency, 1997). The model includes a central exchange compartment, 15 peripheral body compartments, and 3 elimination pools (Figure 4-8). The central exchange compartment is the *diffusible* pool of lead in plasma. The model simulates a bound pool in plasma (i.e., lead bound to plasma proteins); that has an equilibrium ratio (bound:free) of approximately 5. Transport of lead from plasma to tissues is assumed to follow first-order kinetics. The temporal resolution of the model is 1 day. Transfer rate constants vary with age and blood lead concentration. The latter adjustment accounts for the limited uptake of plasma lead into red blood cells and the resulting shift in distribution of lead from plasma-ECF to other tissues. Above a nonlinear threshold concentration in red blood cells (assumed to be 60 μg/dL), the rate constant for transfer to red blood cells declines and constants to all other tissues increase proportionally (Leggett, 1993). This replicates the nonlinear relationship between plasma and red blood observed in humans (Smith et al., 2002; Manton et al., 2001; Bergdahl et al., 1999, 1998, 1997). The model simulates blood volume as an age-dependent function, which allows simulation of plasma and blood lead concentrations. However, volumes of other tissues are not simulated; therefore, only lead masses in these tissues, and not concentrations are simulated.

First-order transfer coefficients (day-1) between compartments were developed for six age groups, and intermediate age-specific values are obtained by linear interpolation. The total transfer rate from diffusible plasma to all destinations (TPALL) combined is assumed to be 2000 day-1, based on isotope tracer studies in humans receiving lead via injection or inhalation. Values for transfer coefficients from plasma to tissues and tissue compartments are based on measured deposition fractions (DF) or instantaneous fractional outflows of lead between tissues

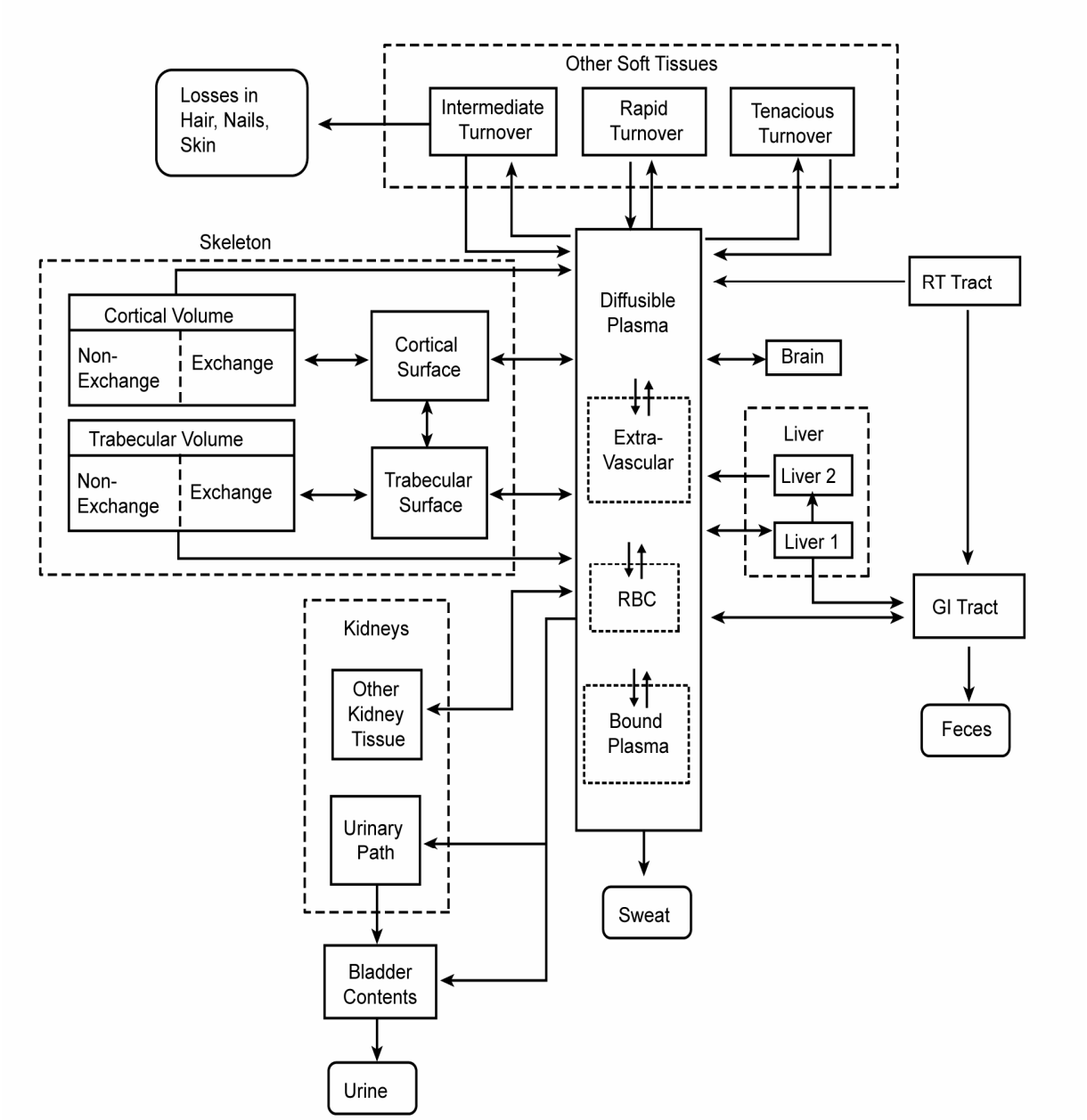


Figure 4-8. Structure of the Leggett Lead Biokinetic Model (Leggett, 1993). The central exchange compartment is *diffusible* plasma. Bone is represented as having surface (which rapidly exchanges with plasma) and volume compartments; the latter stimulates slow exchange with the surface and slow return of lead to the plasma from bone resorption.

compartments (Leggett, 1993), where the transfer coefficient to a specific tissue or compartment (TP_i) is given by:

$$TP_i = DF_i \cdot TPALL \quad (4\text{-}3)$$

This approach establishes mass balance with respect to the transfer rates from plasma:

$$\sum TP_i = TPALL \quad (4\text{-}4)$$

The model simulates both rapid exchange of lead with plasma via bone surface and slow loss by bone resorption. Cortical bone volume (80% of bone volume) and trabecular bone volume (20% of bone volume) are simulated as bone surface compartments, which rapidly exchanges with lead in plasma, and bone volume, within which are *exchangeable* and *nonexchangeable* pools. Lead enters the exchangeable pool of bone volume via the bone surface and can return to the bone surface, or move to the nonexchangeable pool, from where it can return to the plasma only when bone is resorbed. Transfers from plasma to bone surface, return from bone surface to plasma, and bone surface to exchangeable bone volume are assumed to be relatively fast processes (adult $t_{1/2}$ = 3.85, 1.4, and 1.4 days, respectively). Return of lead from the exchangeable bone volume is slower (adult $t_{1/2}$ = 30 days); however, the dominant transfer process determining long-term accrual of bone lead burden are slow rate coefficients for transfer of lead from the nonexchangeable pools of trabecular and cortical bone to plasma (adult $t_{1/2}$ = 3.8 and 23 years, respectively). Bone transfer coefficients vary with age (faster in children) to reflect the age-dependence of bone turnover. The slow, nonexchangeable, bone volume compartment is much more labile in infants and children than in adults (e.g., cortical $t_{1/2}$ = 68 days at birth and 1354 days at age 15 years; trabecular $t_{1/2}$ = 68 days at birth and 725 days at age 15 years). Other physiological states (such as pregnancy and menopause) that affect bone turnover and, therefore, bone lead kinetics are not simulated, although such states could conceivably be accommodated with adjustments to tissue (e.g., bone) transfer coefficients.

The liver is simulated as two compartments; one compartment has a relatively short removal half-life for transfers to plasma and to the small intestine by biliary secretion (adult $t_{1/2}$ = 10 days); a second compartment simulates a more gradual transfer to plasma of

approximately 10% of lead uptake in liver (adult $t_{1/2}$ = 365 days). The kidney is simulated as two compartments, one that exchanges slowly with blood plasma and accounts for lead accumulation kidney tissue (adult $t_{1/2}$ = 365 days) and a second compartment that receives lead from blood plasma and rapidly transfers lead to urine (adult $t_{1/2}$ = 5 days), with essentially no accumulation (urinary pathway). Other soft tissues are simulated as three compartments representing rapid, intermediate, and slow turnover rates, without specific physiologic correlates (adult $t_{1/2}$ = 0.3, 100, and 1824 days, respectively). Other excretory pathways (hair, nails, and skin) are represented as a lumped pathway from the intermediate turnover rate of the soft tissue compartment.

The Leggett model simulates lead intakes from inhalation, ingestion, or intravenous injection. The latter was included to accommodate model evaluations based on intravenous injection studies in humans and animal models. The respiratory tract is simulated as four compartments into which inhaled lead is deposited and absorbed with half-times of 1, 3, 10, and 48 hours. Four percent of the inhaled lead is assumed to be transferred to the GI tract. These parameter values reflect the data on which the model was based, which were derived from studies in which human subjects inhaled submicron lead-bearing particles (Morrow et al., 1980; Chamberlain et al., 1978; Wells et al., 1975; Hursh and Mercer, 1970; Hursh et al., 1969). These assumptions would not necessarily apply for exposures to larger airborne particles. Absorption of ingested lead is simulated as an age-dependent fraction of the ingestion rate, declining from 0.45 at birth to 0.3 at age 1 year (to age 15 years), and to 0.15 after age 25 years (Figure 4-9).

4.4.2 Model Calibration and Evaluation

Leggett (1993) and Pounds and Leggett (1998) describe various qualitative empirical comparisons of model predictions against observations made on adults (e.g., Skerfving et al., 1985; Campbell et al., 1984; Manton and Cook, 1984; Barry, 1981; DeSilva, 1981; Chamberlain et al., 1978; Rabinowtiz et al., 1976; Barry, 1975; Griffin et al., 1975; Gross et al., 1975; Hursh and Mercer, 1970; Hursh et al., 1969; Schroeder and Tipton, 1968). Age-specific changes in parameter values that specify the biokinetics of lead in children were assigned values that resulted in agreement between predicted age-specific lead distribution (fraction of body burden) in blood, bone, brain, kidney, liver, and other tissues, and reported postmortem values (Schroeder and Tipton, 1968; Barry, 1975, Gross et al. 1975; Barry, 1981).

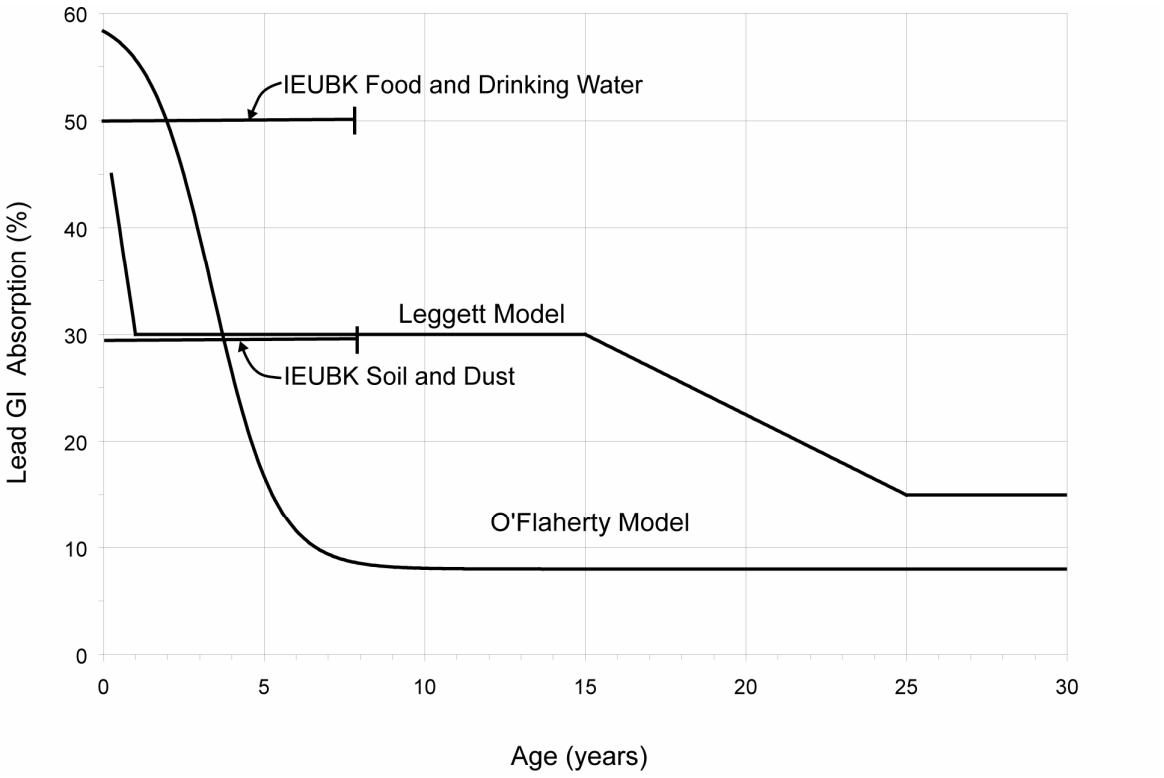


Figure 4-9. Age-dependency of absorption fraction for ingested lead in the Leggett and O'Flaherty models. The IEUBK model projects absorption only through age seven (84 mo). At intakes below those which approach the limit on "active" absorption of lead, absorption is constant with age, with default valves of 50% for diet and drinking water, 30% for soil and dust. Fractional absorption via the active pathway decreases with age and lead intake (see Figure 4-7).

4.4.3 Model Applications

Biomarkers Simulated. The Leggett model simulates the concentrations of lead in blood and plasma, lead masses of lead in bone and various soft tissues, and excretion of lead in urine that correspond to lifetime exposures (in terms of daily lead intakes).

Exposure Inputs. The model does not contain a detailed exposure module (although it can be linked to an exposure model); lead exposure estimates are incorporated into the simulations as age-specific point estimates of average daily intake (µg/day) from ingestion, inhalation, or injection. The model operates with a lead intake time step of 1 day, which allows simulation of rapidly changing (i.e., daily) intermittent exposures (Lorenzana et al., 2005; Khoury and

Diamond, 2003). Assumptions of blood lead concentrations at birth can also be introduced into the simulations, from which levels in other tissue in the first time step after birth are calculated.

Dose reconstruction is possible with this model, since intakes, and corresponding tissue lead burdens accrued at any period in the lifetime, prior to an exposure event of interest, can be simulated. Pounds and Leggett (1998) illustrate this in a study of a childhood lead poisoning case, in which the exposure is followed by chelation. Chelation was simulated as a short-duration increase in the plasma lead deposition fraction to urine, with corresponding proportional decreases in deposition fractions to other tissues.

4.4.4 Implementation Code

The Leggett model was initially developed as a Fortran code, which can be run, without compiling, from various platforms, including DOS and Windows (see Pounds and Leggett, 1998 for a description). A version compiled in Advanced Continuous Simulation Language (ACSL) has also been reported (Lorenzana et al., 2005). Confirmation of the Leggett model code was carried out by a panel of experts (ICRP, 1989, 1993).

4.5 O'FLAHERTY MODEL

The O'Flaherty model simulates lead exposure, uptake, and disposition in humans, from birth through adulthood (O'Flaherty, 1995, 1993). Figure 4-10 shows a conceptualized representation of the model. Important novel features of the O'Flaherty model are the simulation of growth, bone formation, and resorption. A growth curve is simulated with a logistic expression relating body weight to age in males or females. The full expression relating weight to age has five parameters (constants), so that it can readily be adapted to fit a range of standardized growth curves for males and females. Tissue growth and volumes are linked to body weight; this provides explicit modeling of lead concentrations in all tissues simulated. Other physiologic functions (e.g., bone formation) are linked to body weight, age, or to both. The model can be implemented with a temporal resolution of 1 day; however, as originally configured, the rate parameters are expressed in time units of years.

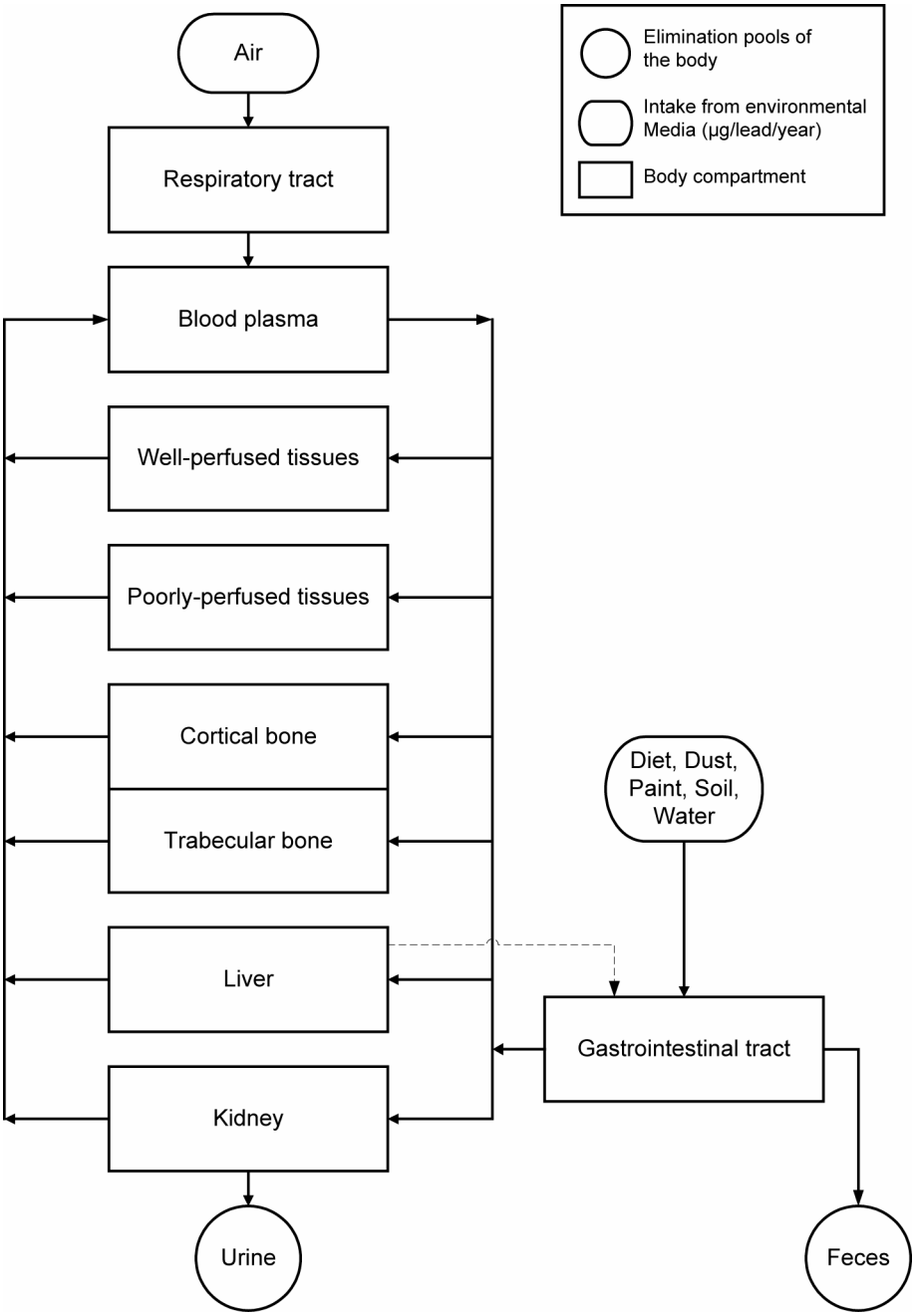
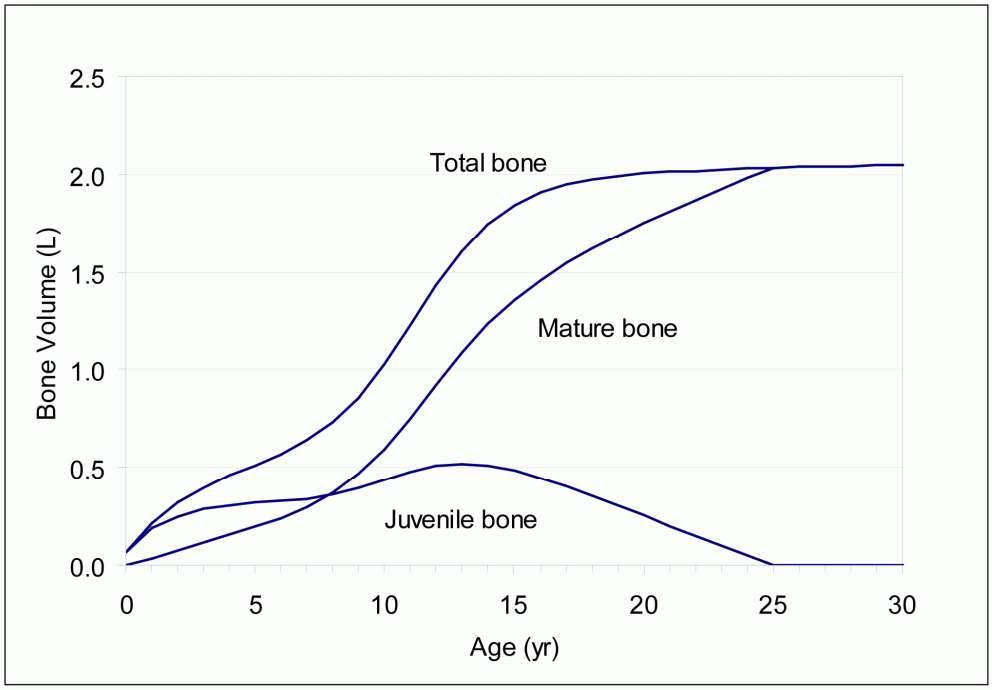


Figure 4-10. Structure of the O'Flaherty Lead Exposure Biokinetics Model (O'Flaherty, 1993, 1995). The central exchanges compartment is *diffusible* plasma. Lead distribution is represented by flows from blood plasma to liver, kidney, richly-perfused tissues, poorly-perfused tissues, and cortical and trabecular bone. The model simulates tissue growth with age, including growth and resorption of bone mineral.

Rates of bone formation and resorption are simulated as age-dependent functions (Figure 4-11). Uptake and release of lead from trabecular bone and metabolically active cortical bone are functions of bone formation and resorption rates, respectively; this establishes the age-dependence to the lead kinetics in and out of bone. Lead exchange between blood plasma and bone is simulated as parallel processes occurring in cortical (80% of bone volume) and trabecular bone (20% of bone volume). The model simulates an age-related transition from immature bone, for which bone turnover (formation and resorption) rates are relatively high, to mature bone, for which turnover is relatively slow. Changes in bone mineral turnover associated with senescence (e.g., postmenopausal osteoporosis) are not represented in the model. Metabolically active regions of bone, in which lead uptake and loss is dominated by bone formation and loss, a region of slow kinetics in mature cortical bone is also simulated, in which lead uptake and release to blood occur by heteroionic exchange with other minerals (e.g., calcium). Heteroionic exchange is simulated as a radial diffusion in bone volume of the osteon. All three processes are linked to body weight, or the rate of change of weight with age. This approach allows for explicit simulation of the effects of bone formation (e.g., growth) and loss, changes in bone volume, and bone maturation on lead uptake and release from bone. Exchanges of lead between blood plasma and soft tissues (e.g., kidney and liver) are represented as flow-limited processes. The model simulates saturable binding of lead in erythrocytes (maximum capacity is 2.7 mg Pb/L cell volume); this replicates the curvilinear relationship between plasma and erythrocyte lead concentrations observed in humans (Smith et al., 2002; Manton et al., 2001; Bergdahl et al., 1999, 1998, 1997). Excretory routes include kidney to urine and liver to bile. Total excretion (clearance from plasma attributable to bile and urine) is simulated as a function of age-dependent glomerular filtration rate. Biliary and urinary excretory rates are proportioned as 70 and 30% of the total plasma clearance, respectively.

The O'Flaherty model simulates lead intake from inhalation and ingestion. Inhalation rates are age-dependent. Absorption of inhaled lead is simulated as a fraction (0.5) of the amount inhaled and is independent of age. Gastrointestinal absorption of lead in diet and drinking water is simulated as an age-dependent fraction, declining from 0.58 of the ingestion rate at birth to 0.08 after age 8 years (Figure 4-9). These values can be factored to account for relative bioavailability when applied to absorption of lead ingested in dust or soil.



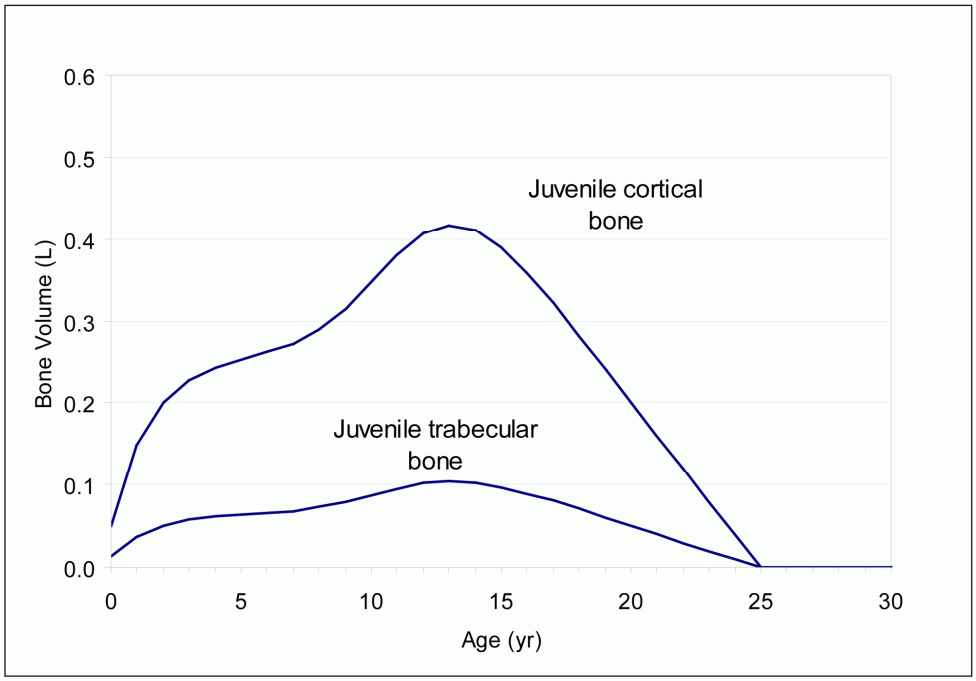


Figure 4-11. Bone growth as simulated by the O'Flaherty Lead Exposure Biokinetics Model (O'Flaherty, 1993, 1995). The model simulates an age-related transition from juvenile bone, in which bone turn-over (formation and resorption) rates are relatively high, to mature bone, in which turn-over is relatively slow. Cortical bone comprises approximately 80% of total bone volume.

4.5.1 Model Calibration and Evaluation

The O'Flaherty model was initially calibrated to predict blood, bone, and tissue lead concentrations in rats (O'Flaherty, 1991a,b,c), and subsequently modified to reflect anatomical and physiological characteristics in children (O'Flaherty, 1995), adults (O'Flaherty, 1993), and Cynomolgus monkeys (*M. fasicularis*) (O'Flaherty et al., 1998). Model parameters were modified to correspond with available information on species- and age-specific anatomy and physiological processes. Empirical comparisons (largely qualitative) of model predictions against observations made in adults (e.g., Van De Vyver et al., 1988; Kehoe, 1987; Marcus, 1985c; Manton and Malloy, 1983; Sherlock et al., 1982; DeSilva, 1981; Moore et al., 1977; Cools et al., 1976; Rabinowitz et al., 1976; Azar et al., 1975) are provided in O'Flaherty (1993); and comparisons against observations made in children (e.g., Sherlock and Quinn, 1986; Bornschein et al., 1985; Chisolm et al., 1985; Lacey et al., 1985) are described in O'Flaherty (1995). Additional discussion of model evaluation can be found in O'Flaherty (1998).

4.5.2 Model Applications

Biomarkers Simulated. The O'Flaherty model simulates lead concentrations in blood and plasma, bone, and various soft tissues, and excretion of lead in urine that correspond to lifetime exposures (in terms of daily lead intakes). Lead in feces is a mixture of unknown proportions of unabsorbed lead in food, drinking water, ingested dust, a small amount of inhaled lead entering the GI tract by the mucociliary clearance from the respiratory tract, and a small amount of absorbed lead eliminated with the red blood cells passing along the bile duct to the GI tract. In this respect, lead in feces represents a poorly defined measure of lead exposure.

Lead in perspiration represents lead in extracellular plasma, but the concentration is low and difficult to measure in a small volume (1 drop ≅ 0.05 mL), and is potentially contaminated with lead in dust on the skin surface.

The model predicts blood lead concentrations for a broad age range (infants to adults), which allows for simulated dose reconstruction, since intakes and corresponding tissue lead burdens accrued at any period in the lifetime, prior to an exposure event of interest can be simulated. Physiological states (such as pregnancy and menopause) that affect bone turnover and, therefore, bone lead kinetics are not simulated, although such states could be accommodated with adjustments to the physiological bone formation and resorption rates.

Exposure Inputs. The O'Flaherty model simulates lead intake by inhalation and ingestion. The model simulates ingestion exposures from infant formula, soil, dust, and drinking water. Rates of soil and dust ingestion are age-dependent, increasing to approximately 130 mg/day at age 2 years, and declining to <1 mg/day after age 10 years. However, the ACSL implementation code allows constructions of simulations with an exposure time step as small as 1 day, which would allow simulation of rapidly changing intermittent exposures (e.g., an acute exposure event).

Modeling Variability and Uncertainty. The O'Flaherty model, as described in O'Flaherty (1995, 1993), utilizes point estimates for parameter values and yields point estimates as output; however, a subsequent elaboration of the model has been reported that utilized a Monte Carlo approach to simulate variability in exposure, absorption, and erythrocyte lead binding capacity (Beck et al., 2001). This approach could be used to predict the probability that children exposed to lead in environmental media will have blood lead concentrations exceeding a health-based level of concern (e.g., 10 μg/dL).

4.5.3 Verification/Validation of O'Flaherty Model

The O'Flaherty model was developed in ACSL. A compiled C program has also been developed (personal communication, E. O'Flaherty). The extent to which code verification and validation studies have been conducted for the O'Flaherty model is unclear at this time. However, analogs of certain components of the O'Flaherty model (e.g., parameters related to bone growth) have been incorporated into the EPA All Ages Lead Model (see Section 4.6) as a potential option for evaluation.

4.6 EPA ALL AGES LEAD MODEL

4.6.1 Model Structure

The EPA All Ages Lead Model (AALM) (Figure 4-12), currently under development, simulates lifetime lead exposures and biokinetics in humans. The model can be used to simulate exposure and biokinetics of lead from birth to age 90 years and is expected to incorporate, at some near-future time, a pregnancy module that simulates transplacental transfer of lead from the other to the fetus.

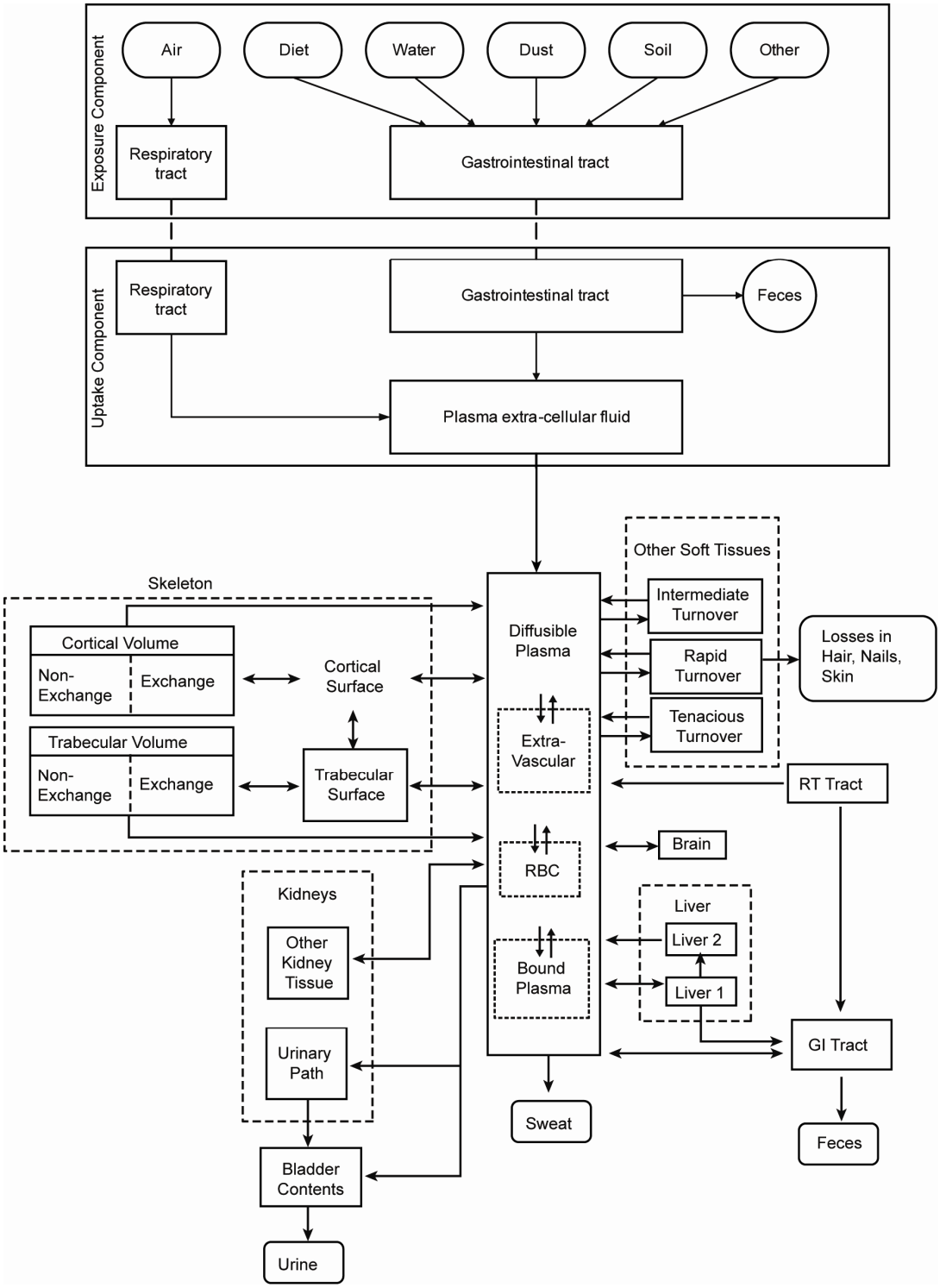


Figure 4-12. Structure of the All Ages Lead Model. The AALM adds a comprehensive exposure component and an uptake component to a revised and recoded version of the Leggett model to produce a model with fully selectable exposure, uptake, and biokinetic parameters.

Exposure Module. The exposure component of the AALM incorporates and extends the exposure component of the IEUBK model. The AALM exposure model defines an individual in terms of age, sex, date of birth, and activity pattern profile. The age specification establishes up to nine age ranges (e.g., infant, child, adolescent, adult, etc.) for which various exposure (and biokinetic) parameter values can be applied. This provides a means for varying parameter values with age. The sex specification links the modeled individual to the appropriate growth algorithm (O'Flaherty 1993, 1995), and the date specification links the individual to historical exposure levels (e.g., air, diet) for the selected age range. The activity pattern specification sets the relative amount of time the individual spends in various exposure settings (e.g., residential, school, recreational, occupational) for which exposure concentrations can be specified. Parameters that can be set include sleep (hours/day); hours/week spent at the residential, school, occupational, and recreational settings; and fraction of time spent indoors at each setting.

The exposure module simulates an average daily intake of lead (μg/day) based on user defined exposure levels in air, diet, dust (ingestion and dermal), drinking water, soil, or paint chips (pica scenario). The air exposure module allows inputs for exposure levels in terms of (a) outdoor air and (b) indoor residential, school, or occupational air, all as a fraction of the outdoor air lead concentration as specified. Ventilation rates (i.e., m^3 air inhaled/day) can be varied for each exposure scenario (e.g., residential, recreational, occupational). The model calculates a weighted average amount of lead inhaled (μg/day) for the combined scenarios and passes this intake rate to the biokinetic model.

The diet exposure module allows input values (current or historical) for lead levels (μg/g) in market basket fruits, vegetables, meat and fish; recreational- or subsistence-harvested fish and meat; and corresponding food intakes for each food type (μg food/day). Lead intake from drinking water is calculated from concentrations (μg/L) in tap water (first draw and/or flushed), fountain water, and/or bottled water; and corresponding source water intake rates (L/day).

The dust exposure module accepts input values for dust concentrations (μg/g) in various settings (e.g., residential, school, recreational, occupational) or dust loadings (μg/m^2) and corresponding dust ingestion rates (μg dust/day) or contact rates (m^2/day), the lead ingestion rate for a given loading being calculated as the product of loading and contact rate. Pica ingestion for soil and/or paint chips can be simulated with input values for lead levels in soil (μg/g) or paint (μg/cm^2) and corresponding pica ingestion rates (g soil/day, cm^2 paint/day). Dermal exposure to

lead in dust can also be simulated with input values for dust lead level (μg/g), dust loading on the skin (mg/cm^2), and skin exposure rate (cm^2/day).

Calculated lead intakes for each exposure pathway are summed to calculate total intakes (μg/day) to the respiratory tract, gastrointestinal tract, and dermal pathway, respectively. The exposure model time step is 1 day (the smallest time interval for a single exposure event).

Biokinetics Module. The biokinetics module of the AALM is based on Leggett (1993) with the following modifications and enhancements (see Figure 4-8 for diagram of the Leggett model):

1. A simulation of dermal absorption is implemented that calculates transfer of lead from the skin to the central plasma compartment, as a function of rate of dermal contact with lead (μg/day) and a dermal absorption fraction.

2. Male and female growth algorithms for body weight, soft tissues, and cortical and trabecular bone are implemented, based on O'Flaherty (1995, 1993). This allows simulation of tissue growth and volumes, as well as lead concentrations in all tissues simulated.

3. A simulation of maternal-fetal transfer is implemented that simulates lead levels in fetal tissues, and establishes blood and tissue lead levels for a postnatal simulation. This provides a means for multigeneration simulation of exposure and lead biokinetics.

4.6.2 Model Calibration and Evaluation

The AALM currently under development incorporates key exposure model features from the IEUBK model (plus age-related extensions) and key biokinetic model features from the Leggett model. To the extent that model validation evaluations have indicated reasonably good matches between IEUBK or Leggett model outputs and empirical observations, the same can be reasonably expected for the AALM. However, this remains to be verified by future AALM model validation evaluations that include comparisons of AALM model run results with other model outputs and with empirical observations.

4.6.3 Model Applications

Biomarkers Simulated. The AALM simulates the concentrations of lead in blood and plasma, bone, and various soft tissues, and excretion of lead in urine that correspond to lifetime exposures, in terms of daily lead intakes. Algorithms for transplacental transfer of lead are also planned to be incorporated soon, so that concentrations of lead in fetal tissue resulting from maternal lead exposures can be simulated.

Exposure Inputs. The model simulates daily lead intakes based on inputted current or historic levels of lead in environmental media (e.g., air, diet, drinking water, dust, soil) in various exposure settings (e.g., residential, school, recreational, occupational). The model operates with a lead intake time step of 1 day, which allows simulation of rapidly changing (i.e., daily) intermittent exposures.

Multigeneration dose reconstruction should be possible with this model, since intakes, and corresponding lead burdens accrued at any period in the lifetime, prior to an exposure event of interest, can be simulated, including lead burdens received in utero.

4.6.4 Validation and Verification of AALM Implementation Code

The AALM is implemented as a compiled C++ program. Code verification and validation are performed with each iteration of the model code as part of the model development process, and these are tracked by a formal tracking process.

4.7 SLOPE FACTOR MODELS

Slope factor models have been used as simpler alternatives to compartmental models for predicting blood lead concentrations, or the change in blood lead concentration associated with a change in exposure (Maddaloni et al., 2005; SRC, 2003b; Abadin and Wheeler, 1997; Stern, 1996; Bowers et al., 1994; Stern, 1994; Carlisle and Wade, 1992). In slope factor models, lead biokinetics are represented as a linear function between the blood lead concentration and either lead uptake (uptake slope factor, USF) or lead intake (intake slope factor, ISK). The models take the general mathematical forms:

$$PbB = E \cdot ISF \quad (4\text{-}5)$$

$$PbB = E \cdot AF \cdot USF \quad (4\text{-}6)$$

where PbB is the blood lead concentration, E is an expression for exposure (e.g., soil intake x soil lead concentration) and AF is the absorption fraction for lead in the specific exposure medium of interest. Intake slope factors are based on ingested rather than absorbed lead and, therefore, integrate both absorption and biokinetics into a single slope factor, whereas models that utilize an uptake slope factor include a separate absorption parameter. In general, slope factor models predict quasi-steady state blood lead concentrations that correspond to average daily lead intakes (or uptakes) that occur over sufficiently long periods to produce a quasi-steady state (i.e., >75 days, ~3 times the $t_{1/2}$ for elimination of lead in blood).

Slope factors used in various models were presented in Table 4-1. Of the models presented in Table 4-2, Bowers et al. (1994) and SRC (2003) implement uptake slope factors. The slope factors used in both models (~0.4 μg/dL per μg Pb/day) are similar to biokinetic slope factors predicted from the O'Flaherty model (0.65 μg/dL per μg Pb uptake/day) and Leggett model (0.43 μg/dL per μg Pb uptake/day) for simulations of adult exposures (Maddaloni et al., 2005). A review of reported intake slope factors relating medium-specific exposures and blood lead concentrations derived from epidemiologic studies can be found in the 1986 AQCD and in Abadin and Wheeler (1997).

4.8 MODEL COMPARISONS

Table 4-2 summarizes the major features of various models of human exposure that predict tissue lead burdens. The slope factor models give similar predictions of quasi-steady state blood lead concentration when similar inputs and parameter values were applied to each model (Maddaloni et al., 2005).

Lead uptake-blood lead concentration relationships in children, predicted by the IEUBK, Leggett, and O'Flaherty models are shown in Figure 4-13. In the range of uptakes shown (0.1-100 μg lead absorbed/day), nonlinearity of the relationship is apparent in the Leggett

Table 4-2. Summary of Models of Human Exposure that Predict Tissue Distribution of Lead

Model	Age Range	Exposure Pathways	Exposure Time Step	Biokinetics Simulation	Biomarkers Predicted	Variability and Uncertainty Simulation
U.S. Environmental Protection Agency IEUBK Model White et al. (1998)	0-7 yr	Air Diet Soil/dust Water Other	1 year	Multicompartmental	Blood lead	Variability: blood lead GSD_i Variability/uncertainty: MCA (Griffin et al., 1999b)
U.S. Environmental Protection Agency AALM (2005)	0-90 yr	Air Diet Soil/dust Water Other	1 day	Multicompartmental	Blood Bone Brain Fetus Kidney Liver Urine	Variability and uncertainty determined by independent assessment of multiple runs of the model.
Leggett (1985)	0-Adult	Intakes (inhaled, ingested, injected)	1 day	Multicompartmental	Blood Bone Brain Kidney Liver Urine	NA
O'Flaherty (1993, 1995)	0-Adult	Air Diet Soil/dust Water Other	1 year (code supports 1 day)	Multicompartmental	Blood Bone Brain Kidney Liver Urine	Beck et al. (2001)

Table 4-2 (cont'd). Summary of Models of Human Exposure that Predict Tissue Distribution of Lead

Model	Age Range	Exposure Pathways	Exposure Time Step	Biokinetics Simulation	Biomarkers Predicted	Variability and Uncertainty Simulation
U.S. Environmental Protection Agency ALM Maddaloni et al. (2005)	Adult	Soil (supports other pathways)	>3 months (quasi-steady state)	Uptake slope factor	Blood	Variability: blood lead GSD_i
California Environmental Protection Agency, Carlisle and Wade (1992)	Child Adult	Air Diet Soil/dust Water	>3 months (quasi-steady state)	Intake slope factor	Blood	Variability: blood lead GSD_i
Bowers et al. (1994)	Adult	Air Soil/dust Water	<3 months (quasi-steady state)	Uptake slope factor	Blood	Variability: blood lead GSD
Stern (1994, 1996)	Child Adult	Dust/soil	>3 months (quasi-steady state)	Intake slope factor	Blood	Variability: blood lead GSD_i: MCA

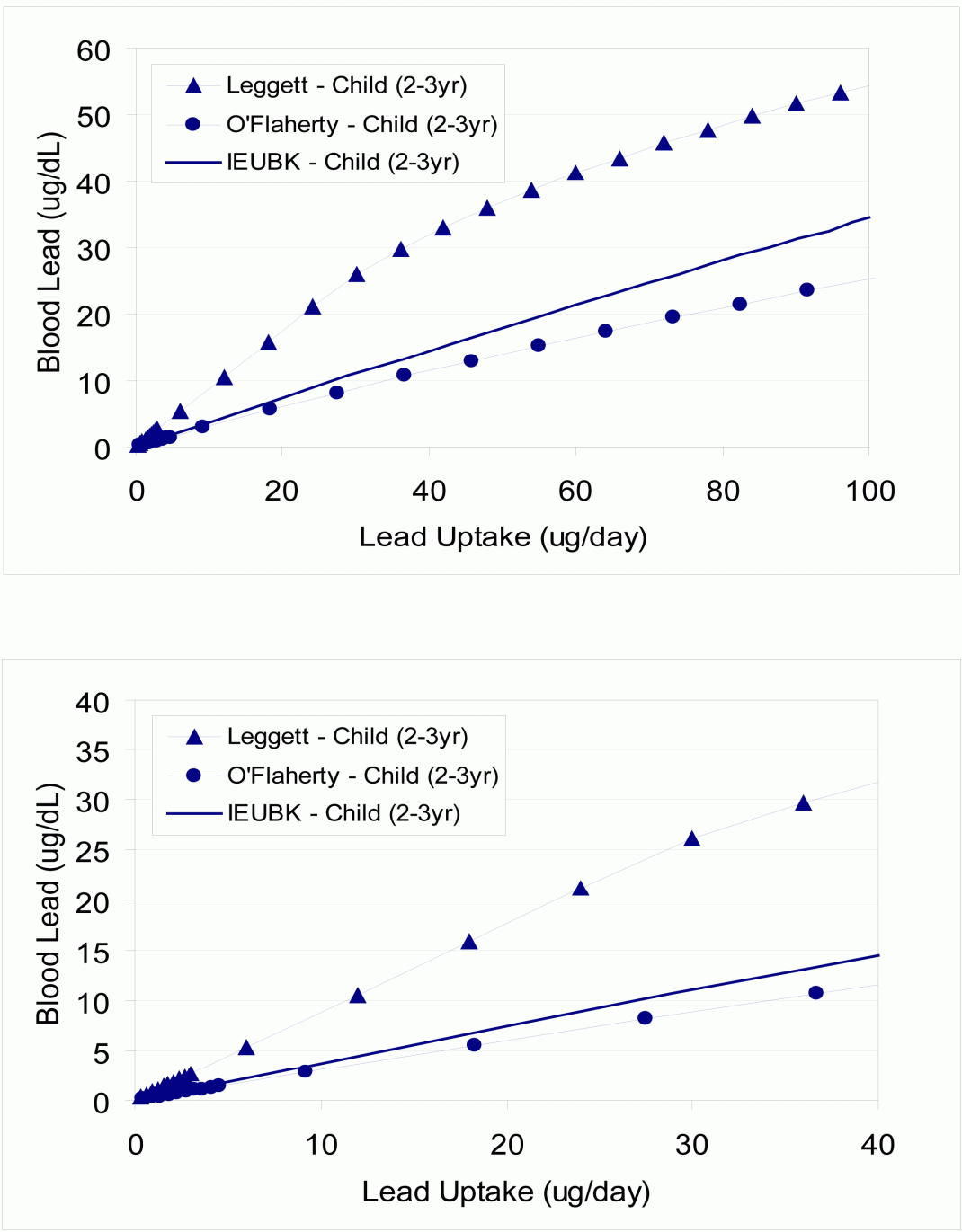


Figure 4-13. Model comparison of predicted lead uptake–blood lead concentration relationship in children. In the range of uptakes shown, the nonlinearity of the relationship is apparent in the Leggett and O'Flaherty Models simulations, reflecting the simulation of the limited capacity of red blood cells to take up lead. Regression slopes (µg /dL blood per µg/day uptake) for the predictions ≤10µg/dL are: Leggett Model, 0.88; IEUBK Model, 0.36; O'Flaherty Model, 0.29.

and O'Flaherty models simulations. This reflects assumptions in each model regarding the limited capacity of red blood cells to take up lead. Regression slopes (μg/dL blood per μg/day uptake) for the predictions ≤10 μg/dL are: Leggett model, 0.88; IEUBK model, 0.36; O'Flaherty model, 0.29. The models predict an average blood lead concentration of 10 μg/dL for the age range 2-3 years, in association with average lead uptakes (μg/day) for the same period of approximately: Leggett model, 12; IEUBK model, 29; O'Flaherty model, 36.

A similar comparison of uptake-blood lead concentration relationships predicted in adults is shown in Figure 4-14. Regression slopes for adults predicted by the Leggett and O'Flaherty models (at blood lead concentrations ≤10 μg/dL) are more similar for adults (Leggett model, 0.54; O'Flaherty model, 0.72) than for children (see Figure 4-13 vs. Figure 4-1). The models predict an average blood lead concentration of 10 μg/dL for the age range 31–32 years, in association with average lead uptakes, for the same period, of approximately 18 and 13 μg/day, Leggett and O'Flaherty models, respectively. The nonlinearity in both children and adults is due largely to the limited capacity of red blood cells to take up lead at concentrations above 15-20 μg/dL. Figure 4-12 shows that the Leggett and O'Flaherty models reach this point at about 30 μg Pb/day for children and about 40 μg/day for adults. The IEUBK model (for children) does not include this nonlinearity feature.

Comparisons of predicted bone and soft tissue lead burdens are shown in Figure 4-15. Leggett and O'Flaherty models predict bone lead burdens. Both the Leggett and O'Flaherty models predict a bone lead burden in adults of approximately 90 and 98% of total body burden, respectively. Regression slopes (mg lead in bone per μg uptake/day) are 1.2 for the Leggett model and 2.1 for the O'Flaherty model.

Figures 4-16 and 4-17 compare model predictions for blood lead concentration for hypothetical childhood or adult lead exposures. The hypothetical child (Figure 4-16) has a blood lead concentration of 2 μg/dL at age 2 years and then experiences a 1-year exposure to 100 μg Pb/day. All three models (Leggett, IEUBK, and O'Flaherty) predict a similar temporal pattern of increase in blood lead concentration at the start of exposure, then attainment of a quasi-steady state, followed by a decrease in blood lead concentration, with fast and slower phases of the decline in blood lead concentration after the exposure ceases. However, differences in the predicted kinetics of the blood lead changes and the predicted quasi-steady state blood lead concentrations are evident. For this hypothetical scenario, the Leggett model

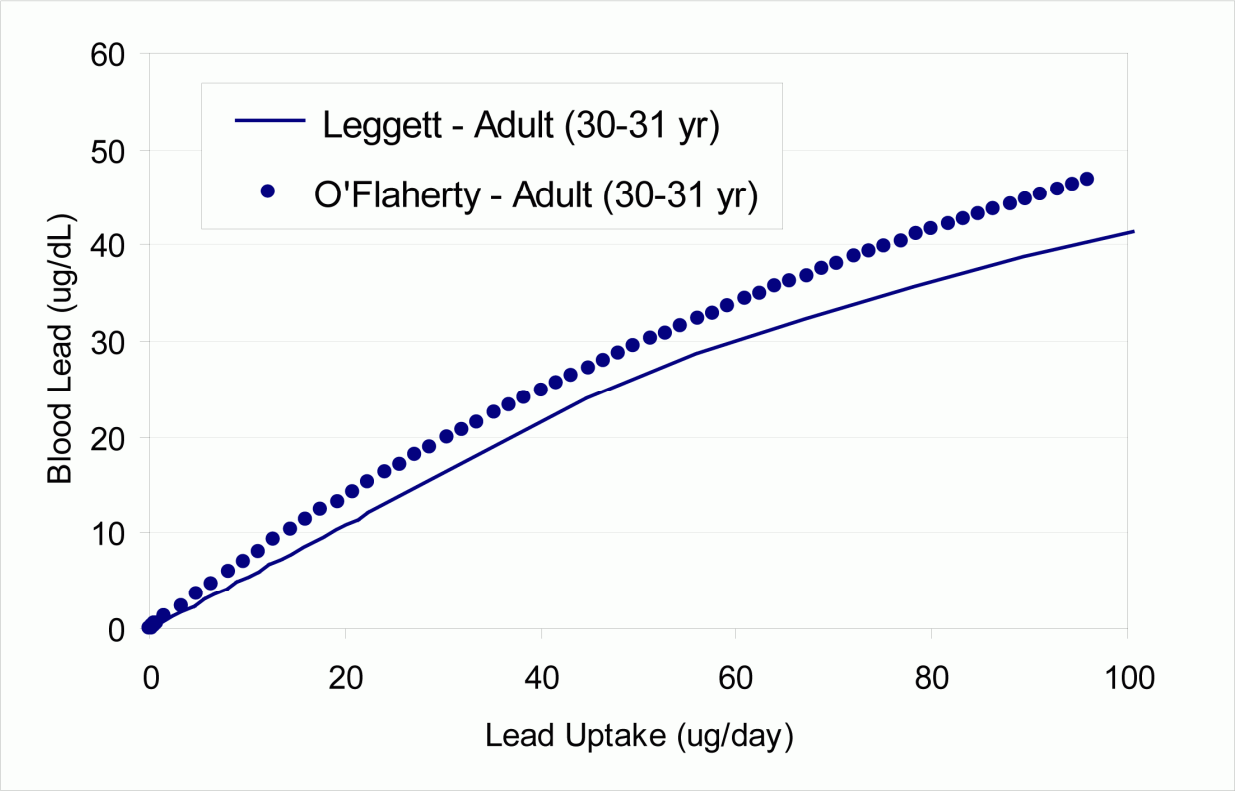


Figure 4-14. Model comparison of predicted lead uptake-blood lead concentration relationships in adults. The nonlinearity of the relationship is apparent in both the Leggett and O'Flaherty Models. Regression slopes (µg /dL blood per µg/day uptake) for the predictions ≤10µg/dL are: Leggett Model, 0.54; O'Flaherty Model, 0.72.

predicts the highest blood lead concentrations (23 µg/dL) compared to the O'Flaherty (12 µg/dL) and IEUBK (10 µg/dL) models. These differences are not solely the result of different values for the absorption fraction in 2–3 year old children (Figure 4-9): Leggett model, 30%; O'Flaherty model, 45% (descending from 49% at age 2 years to 39% at age 3 years); IEUBK model, 25% (at a soil lead intake of 100 µg/day). A similar pattern is evident in the simulation of the same exposure (100 µg/day for 1 year) in an adult (age 30 years; Figure 4-17). The Leggett model predicts a quasi-steady state blood lead concentration of approximately 8.2 µg/dL and the O'Flaherty model predicts 5.4 µg/dL. However, most of this difference can be attributed to the different absorption fraction values used for adults in the two models; 15% in the Leggett model and 8% in the O'Flaherty model.

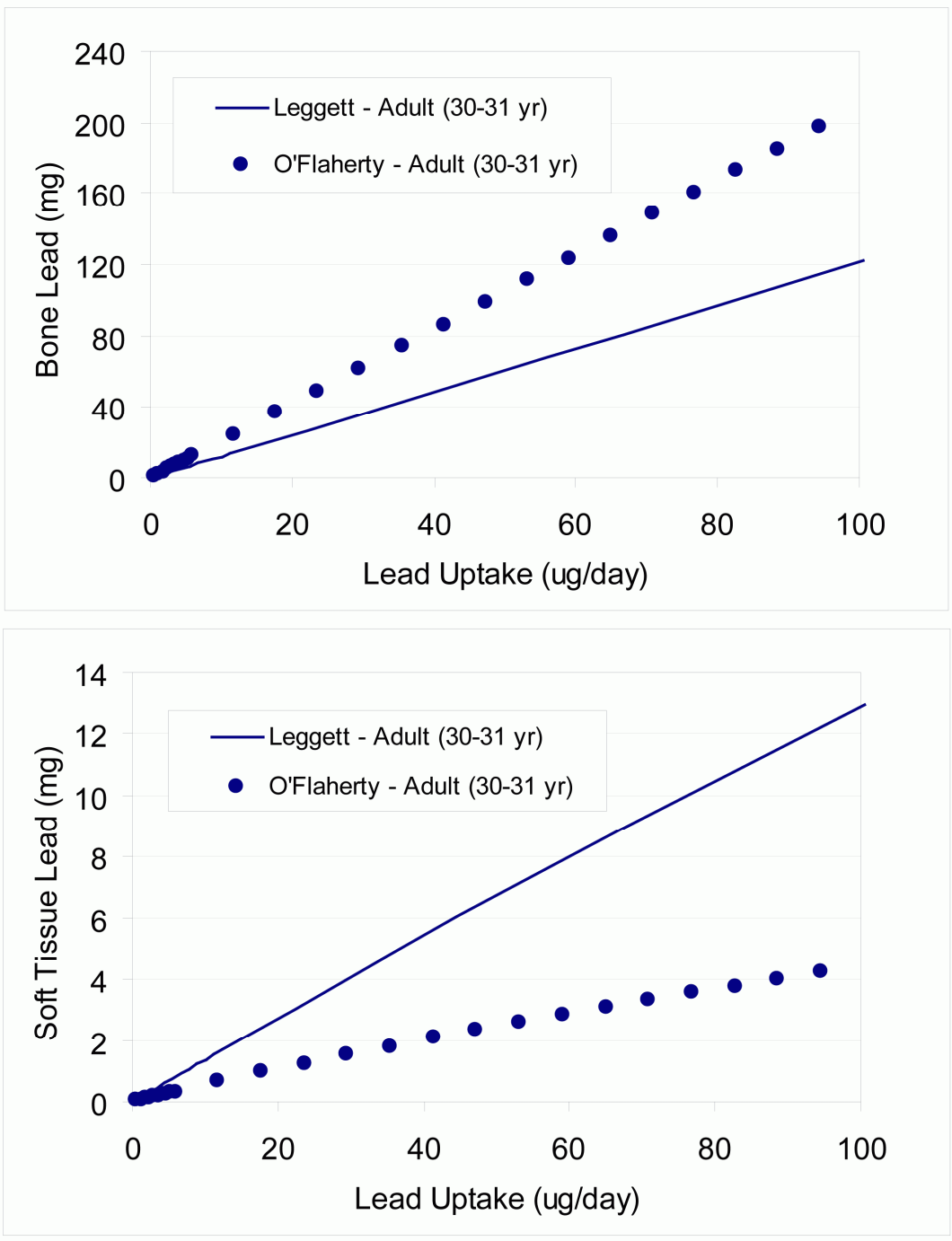


Figure 4-15. Model comparison of predicted of lead uptake-bone and soft tissue lead burden relationship in adults. Both the Leggett and O'Flaherty Models predict a bone lead burden of approximately 90% and 98% of total body burden, respectively. Soft tissue burdens shown include blood. Regression slopes (mg Pb per µg uptake/day) for uptake-bone burden relationship is: Leggett, 1.2; O'Flaherty Model, 2.1.

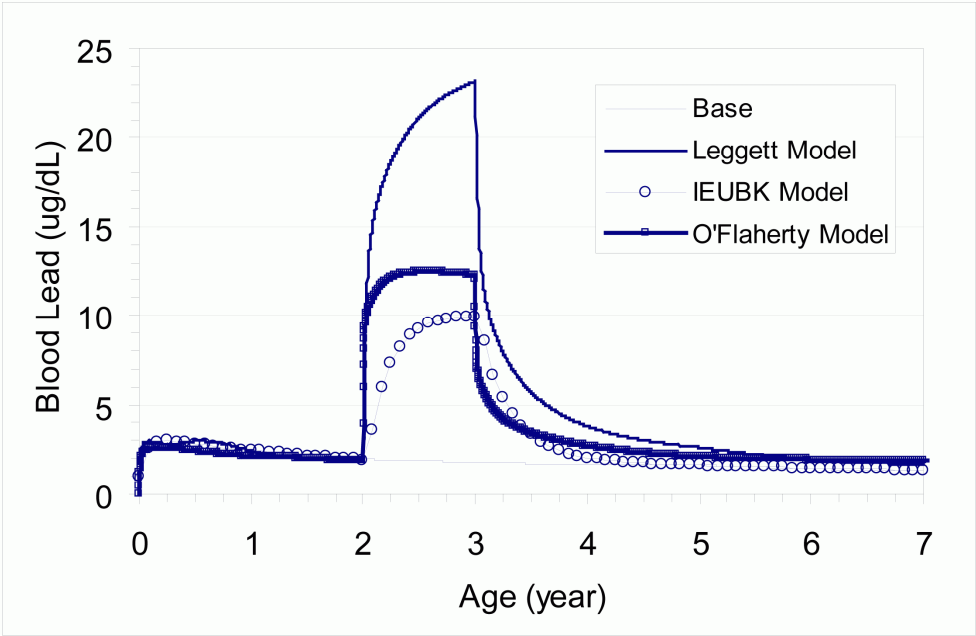


Figure 4-16. Comparison of model predictions for childhood lead exposure. The simulations are of a hypothetical child who has a blood lead concentration of 2 µg/dL at age 2 years, and then experiences a 1-year exposure to 100 µg Pb/day. Default bioavailability assumptions were applied in all three models.

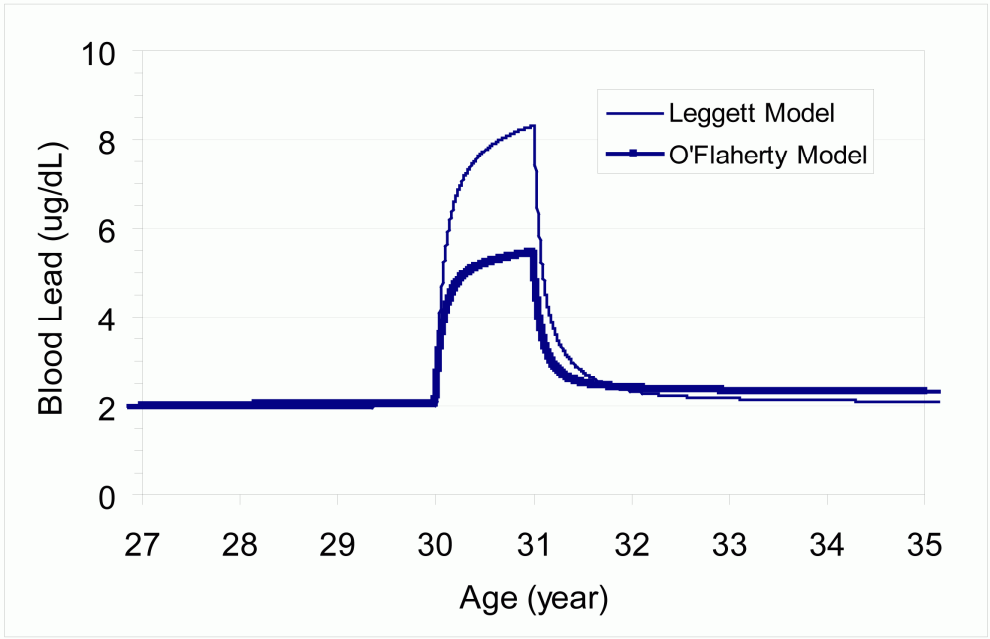


Figure 4-17. Comparison of model predictions for adult lead exposure. The simulations are a hypothetical adult who has a blood lead concentration of 2 µg/dL at age 30 years and then experiences a 1-year exposure to 100 µg Pb/day. Default bioavailability assumptions were applied in the Leggett and O'Flaherty models.

4.9 CONCLUSIONS AND FUTURE DIRECTIONS

Modeling of relationships between lead exposures and lead levels in tissues has advanced considerably during the past 25 years or so. Three multicompartmental exposure-biokinetics models have been developed and evaluated to varying degrees for predicting associations between exposure and body burden (IEUBK model, Leggett model, O'Flaherty model). A fourth model, the All Ages Lead Model (AALM), is still under development and may resolve some of the issues regarding minor discrepancies between other models, while at the same time adding new features directly applicable to risk assessment.

The IEUBK model has had the most extensive application in the regulatory context, as EPA guidance recommends that, where possible, risk estimates for residential exposures to lead at hazardous waste sites be based on IEUBK model predictions of blood lead concentrations in children. Although, these models are constructed very differently (e.g., the O'Flaherty biokinetics model has only 17 lead parameters, compared to 65 in the Leggett biokinetics model, and 47 in the IEUBK biokinetics model), the three models yield remarkably similar predictions of blood lead concentration for similar hypothetical exposure scenarios. The three models predict similar kinetics of change of blood lead concentrations in association with a change in lead exposure (e.g., Figures 4-16 and 4-17). Both the Leggett and O'Flaherty models predict similar rates of lead accumulation in bone, for the same rates of uptake of lead into the body. Predictions of quasi-steady state blood lead concentrations for the scenarios are simulated in Figures 4-16 and 4-17 and differ across models by a factor of approximately 2. While this magnitude of difference may be substantial in the context of regulatory use of the models (e.g., for establishing cleanup goals at hazardous waste sites), it represents a remarkable convergence of various approaches taken to reduce the complex biokinetics of lead to tractable, and relatively simple, mathematical expressions. Given that the AALM incorporates and combines key features from predecessor models (especially exposure components of the IEUBK and the Leggett biokinetics components) it is reasonable to expect likely convergence of its outputs with those of such predecessor models, but this remains to be verified by future validation evaluations for the AALM.

Several major challenges remain to be confronted in further developing our ability to simulate lead exposure-tissue level relationships in real individuals or populations. The three earlier mechanistic models described above do not simulate the kinetics of lead in pregnancy or

in senescence (e.g., menopause). Only one of these three earlier models (Leggett) simulates lead levels in brain, a potential target organ for lead toxicity. None of the models have been rigorously evaluated for accuracy of predictions of bone lead levels in humans, for which there is a rapidly expanding set of observations of importance to dose-response assessment. The fourth multicompartmental model discussed above, the EPA AALM currently under development simulates lead kinetics out to age 90 years and include features designed to simulate both maternal and fetal lead biokinetics during pregnancy (as a future option). In addition to these useful features of the biokinetics model, the AALM incorporates a life-time exposure module to simulate complex life-time exposure patterns (i.e., complex temporal patterns of exposure to multiple exposure media and in multiple exposure settings). These exposure simulations can be used to drive simulations of lead biokinetics, based on the AALM biokinetics model or other lead biokinetics models.

While extending the functionality of the models, as noted above, the AALM also provides important insight with regard to desirable future directions in model development. Of great importance for regulatory uses of the models, for example, is the need for more rigorous quantitative assessment of confidence (i.e., uncertainty) in model predictions. To date, such assessments have not been applied uniformly in a manner that allows cross-model comparisons of confidence for specific regulatory uses.

The IEUBK Model has undergone the most extensive and thoroughly reported evaluation of a regulatory use of the model, i.e., (a) quantitative evaluation of predicted distributions of blood lead concentrations in children who live in areas for which cross-sectional measurements of environmental lead levels were available and (b) independent verification of the IEUBK model implementation code (Hogan et al., 1998; Zaragoza and Hogan, 1998). However, a similar level of evaluation of the Leggett and O'Flaherty models has not been reported, although specific predictions of the models have been evaluated against observations (e.g., experimentally- observed kinetics of change in blood lead following a change in intakes). Nor has the AALM yet undergone a similar level of evaluation as the IEUBK model.

To a large extent, the important information gap regarding evaluation of model confidence derives from a lack of observational data and/or public access to observational data on which predictions could be evaluated. An additional challenge for applications of the models in a regulatory context relates to uncertainties in exposure data from which exposure model inputs

are derived. Model development and uncertainty assessment could be substantively advanced by assembling verified (for accuracy) sets of data on lead biokinetics against which models could be uniformly evaluated. Examples of the types of data that would be valuable include data on the kinetics of change in blood or tissue lead concentrations, or stable lead isotope ratios, in response to a change in exposure. Also, access to large data bases that include reported lead exposure measurements for various media that are paired with blood or tissue lead measurements for individuals affected by pertinent exposure scenarios would also be extremely valuable for cross-model evaluations.

4.10 REFERENCES

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5. TOXICOLOGICAL EFFECTS OF LEAD IN LABORATORY ANIMALS, HUMANS, AND IN VITRO TEST SYSTEMS

5.1 INTRODUCTION

As noted in Chapter 1, air quality criteria documents evaluate scientific knowledge of relationships between pollutant concentrations and their effects on the environment and public health. Chapters 2 and 3 of this document discussed the chemistry and physical properties of lead (Pb); sources, emissions, transport, and deposition of Pb; and environmental concentrations and pathways to human exposure. Chapter 4 discussed models of human exposure that predict tissue distribution of lead. This chapter (Chapter 5) assesses information regarding the toxicological effects of Pb in laboratory animals, humans, and in vitro test systems. Emphasis is placed here on qualitative characterization of various Pb-induced effects, with attempts to define dose-effect relationships for the key health effects that are thought to occur at ambient exposure levels encountered by the general population of the United States. Chapter 6 follows with a discussion of epidemiologic studies of ambient Pb-exposure effects. Chapter 7 provides an integrative synthesis of information on Pb exposures and health effects. The environmental effects of Pb are discussed in Chapter 8.

The framework used here for presenting the toxicologic effects of Pb is subdivided mainly according to organ systems. As noted in the 1986 Pb AQCD, this facilitates presentation of the information, but it must be stressed that all systems are interdependent, functioning in delicate concert to preserve the physiological integrity of the whole organism.

The information discussed in this chapter is derived from a very wide body of literature on studies in humans, laboratory animals, and in vitro test systems of animal cell lines and organ systems that may mimic responses in intact animals. This chapter is not intended to be a compendium of all that is known about lead; rather, it is an update of the reported biological effects from the last previous Pb AQCD (U.S. Environmental Protection Agency, 1986), the Addendum to that document (Lead Effects on Cardiovascular Function, Early Development, and Stature) (U.S. Environmental Protection Agency, 1986), and the Supplement to the 1986 Addendum (U.S. Environmental Protection Agency, 1990). The historical Pb literature is briefly

summarized at the opening of each section or subsection and is intended as a very concise overview of previous work The reader should refer to the previous documents listed above for more detailed discussion of the literature prior to the late 1980s. Each section then continues with brief discussions of key studies published since 1986. Longer discussions of the newly available studies are included where warranted. Sections are ended with comparisons of data from the 1986 AQCD with new data, and basic conclusions are drawn. More detailed summaries of newly available studies and results are provided in tables in Annex AX5.

5.2 EFFECTS OF LEAD ON HEME SYNTHESIS

5.2.1 Effects of Lead on Erythrocyte Biology and Function

Lead poisoning is one of the most common acquired environmental diseases, because of physical properties of the metal and its widespread distribution in the environment. It is a complex disorder affecting several organs in the body, including developing erythrocytes (red blood cells [RBCs]). Anemia is frequently observed with Pb poisoning and is thought to result from the shortening of erythrocyte life span and is also due to the effects of Pb on hemoglobin synthesis. However, the exact mechanisms by which Pb affects the red blood cell (RBC) life span and heme synthesis are not clear. It is postulated that the mechanisms may be due to the effects of Pb on iron uptake; Pb poisoning also causes an increased urinary excretion of porphyrins and 5-aminolevulinic acid (ALA), the first precursor for heme synthesis. In addition, the striking similarities between Pb poisoning and acute intermittent porphyria (the disease associated with lesions in the heme biosynthetic enzyme, porphobilinogen deaminase) strongly suggests that one of the major sites of Pb intoxication is the heme biosynthetic pathway.

The 1986 Pb AQCD presented a concise summary of literature available at that time from both animal and human studies indicating potential effects of Pb intoxication on enzymes and precursors involved in heme synthesis, erythrocyte morphology and function as well as the influence of these perturbations on the nervous system and vitamin D metabolism and associated physiological process. In summary, these studies reported an association between increased Pb exposure and increased ALA-S activity (which is increased in kidney with acute exposure and in spleen with chronic exposure, while it decreased in liver tissue in both the exposure scenarios). The activity of ALA-D appeared to be inversely correlated to blood Pb values and was found to

be inhibited in several tissues. It was also inferred from several animal studies that the effect of Pb on heme formation involved both ferrochelatase inhibition and impaired mitochondrial transport of iron. Human studies indicated that occupational exposure to Pb results in decreased erythrocyte cell survival and alterations in erythrocyte membrane integrity and energetics. The vast scientific literature on the effects of Pb on various aspects of heme metabolism in diverse organ systems both in human and animals has accumulated over the past two decades. Recognizing the magnitude of this literature, this chapter is primarily concerned with discussions of data from animal and in vitro studies, while the human studies are dealt with in Chapter 6.

5.2.2 Effects of Lead on Erythrocyte Functions

The cellular membrane is one of the main targets for toxic effects of heavy metals, including Pb. Anemia, one of the clinical symptoms of Pb intoxication, can develop because of impairment of hemoglobin synthesis and damage of erythrocyte membranes by Pb ions. Although, erythrocyte membrane is not as specialized as other cell membranes are, it carries out important functions common to other cell membranes, such as active and passive transport and the production of ionic and electric gradients. Changes in erythrocyte membrane lipid and protein profiles can alter the membrane fluidity, potentially affecting enzymatic activity and the functionality of receptors and ion channels present on the plasma membrane and also can influence the ionic and molecular composition of intracellular spaces.

Lead Uptake, Binding, and Transport

Studies by Simons (1986a) indicated that the uptake of Pb into human RBCs is a passive process, i.e., it does not require the use of energy in the form of ATP. In addition, Pb may be able to cross the membrane passively in either direction. This process involves anion transport mechanisms, as the characteristic anion exchange inhibitors have been found to inhibit the passive uptake of Pb by RBCs (Simons, 1986a,b). It has also been demonstrated that the transport of Pb across the membrane depends on the presence of another anion, the bicarbonate ion, and is transported as Pb-carbonate (Simons, 1986a). When Pb enters the cell, it binds mainly to hemoglobin, and the ratio of bound to free Pb in cytoplasm has been estimated to be 6000:1. Simons (1986a,b) carried out studies using citrate buffers, which may cause hemolysis of RBCs. To avoid the influence of a citrate buffer, Sugawara et al. (1990) measured the uptake

of Pb into human RBCs by adding Pb directly into plasma. These investigators also found that the transport of Pb across the erythrocyte membrane is energy-independent (passive) and carrier mediated. Little release of Pb from the cells was observed, suggesting absence of any hemolysis of the cells in this protocol. Furthermore, the progressive accumulation of Pb was not observed. More than 98% of the Pb was found accumulated in the cytoplasm in protein-bound form, while only 2% was found in the membrane fraction. Sugawara et al. (1990) also reported finding 45 Pb-binding sites on human hemoglobin. On the other hand, studies reported by Bergdahl et al. (1997) using liquid chromatography coupled with inductive plasma mass spectrometry analysis suggested aminolevulinic acid dehydratase (ALAD), the enzyme involved in the heme synthesis pathway, to be the principle Pb-binding protein, not hemoglobin, as previously thought.

Additional studies carried out by Simons (1993a) evaluated the transport of Pb into RBCs for cell Pb contents in the range of 1 to 10 μM and reported that ^{203}Pb uptake was mediated by an anion exchanger and the efflux was mediated through a vanadate-sensitive pathway identified with the calcium pump (Simons, 1988). He further concluded that the high ratio of RBC to plasma Pb observed in vivo was due to a labile Pb-binding component within the cytoplasm. Simons (1993a) also observed that exit of Pb ions from the RBC was much lower than expected based on his earlier work with erythrocyte ghosts. Utilizing a group of drugs that modify anion exchange and thiol groups in the cytoplasm, Lal et al. (1996) showed that anion exchange mechanisms and thiol groups were critical factors in how Pb stimulates calcium-dependent processes in erythrocytes. Once the role of anion exchanger proteins had been implicated in Pb transport in erythrocytes, Bannon et al. (2000) investigated whether similar anion exchange processes are involved in the uptake and transport of Pb in other cells, such as Madin-Darby canine kidney epithelial cells. Based on a comparative in vitro study using human erythrocytes and canine kidney epithelial cells, these authors reported transport of Pb in kidney epithelial cells, suggesting similar anion exchange involvement.

Erythrocyte Survival, Mobility, and Membrane Integrity

It is well recognized that Pb intoxication interferes with RBC survival by shortening the life span and altering the mobility of the erythrocytes; however, the molecular mechanisms behind these effects of Pb on erythrocyte functions are not well understood. The shape and deformability of the human erythrocyte, or RBC is maintained by several factors including low

concentration of free intracellular Ca^{2+} (<0.1 μM) and a replenished ATP level. An elevated interfacial Ca^{2+} concentration inside the RBC activates the passive ion efflux via a K^+ selective (voltage independent) channel and a concomitant water transport (Gordos effect). Low concentrations of Pb ions can mimic Ca^{2+} and activate the same channel in the RBC.

Intraperitonially injected Pb significantly decreases rat erythrocyte membrane mobility (Terayama et al., 1986), an effect evident to some extent even below blood Pb concentration of 100 μg/100 ml. This decrease in rat erythrocyte mobility was found simultaneous or prior to changes in hematological parameters such as hemoglobin (Hb) levels and hematocrits (Hct). The same group (Terayama and Muratsugu, 1988) also reported a significant decrease in erythrocyte membrane sialic acid content at the same levels of blood Pb with exposure to Pb (20 mM Pb-acetate once a week for 5 weeks). Additional studies by the same group reported that other hematological parameters, such as mean corpuscular volume (MCV), mean corpuscular hemoglobin (MCH), and mean corpuscular hemoglobin concentration (MCHC), were also significantly decreased upon Pb exposure, along with decreased mobility, sialic acid content, and deformability of rat RBCs. It was speculated that Pb-induced decreases in sialic acid content and deformability of RBCs shorten RBC survival time and may lead to anemia in Pb poisoning. Jehan and Motlag (1995) reported Pb exposure caused significant change in RBC membrane cholesterol and phospholipid contents along with sialic acid. Coexposure to Zn was found to reduce these alterations.

Pb-induced morphological changes in human RBC were studied by Eriksson and Bering (1993) using electron paramagnetic resonance imaging. These authors reported that Pb ions (a) induced time-dependent changes in MCV and cell shrinkage and (b) inhibited the Gardos effect. Trialkyl-Pb compounds have also been reported to induce hemolytic activity in erythrocytes, with intensity increasing with hydrophobicity of the compounds (Kleszcynska et al., 1997). Serrani et al. (1997) reported that Pb ions confer protection against RBC lysis in hypotonic low ionic strength media, presumably be due to interaction of Pb with certain constituents in the cell membrane. This resistance to erythrocyte lysis was found to significantly increase with Pb (20 to 25 μM) compared to other metals such as Al, Cd, and Zn (Corchs et al., 2001). The Pb-induced reduction in MCV (RBCs derived from umbilical cord) was found to be reversed when the cells were treated with quinidine, an inhibitor of a potassium channel activator, without any effect on resistance to cell lysis, suggesting changes in cell membrane

structure. This effect may also be involved in membrane deformability (Mojzis and Nistiar, 2001).

Heavy metals, including Cd, Zn, and Pb, have been found to alter RBC membrane microviscosity and fluidity (Amoruso et al., 1987). These authors labeled RBC membranes with fluorescent lipid probe all trans 1, 6-diphenyl-1,3,5-hexatriene (DPH) and demonstrated increased polarization with increased membrane lipid viscosity on exposure to heavy metals. They also postulated that such alterations in cell membrane lipid and possibly also protein fluidity may contribute to abnormal cellular function. Similar changes in RBC fluidity were observed in the RBC collected from workers exposed to Pb (Cook et al., 1987). The RBC ghost membranes isolated from Pb- exposed workers exhibited a significant increase in phosphotidylcholine to phosphotidylethanolamine ratio (an established correlate of membrane fluidity) along with an increase in RBC cholesterol levels, as also reported by Jehan and Motlag (1995) discussed above. These authors predict that such alterations in phospholipid composition of the membrane are responsible in biochemical instability of RBC in Pb-exposed workers. Zimmermann et al. (1993) investigated the potential of such membrane lipid alterations to cause resistance to oxidation. These authors induced hyperlipidemia by treating Pb-exposed Wistar rats with triton. They observed an increase in erythrocyte choline phospholipid levels together with a significant decrease in membrane lipid resistance to oxidation. These authors postulated that such a decrease in resistance might cause RBC fragility, and ultimate destruction, leading to anemic conditions. It has been also reported that exposure to Pb may also increase the levels of fatty acids, e.g., arachidonic acid, in the RBC membrane in humans exposed to Pb (Osterode and Ulberth, 2000). Based on the negative correlation between serum calcium and increased arachidonic acid content, these authors postulated that Pb ions might have substituted for calcium in the activation of phospholipase enzymes, leading to increased synthesis of arachidonic acid. Suwalsky et al. (2003) investigated the interaction of Pb with the RBC membrane, utilizing intact as well as isolated unsealed RBC membrane models (representing phospholipids present in the inner and outer layers of the membrane). Electron microscopy, fluorescence spectroscopy, and X-ray diffraction analyses of these models by the authors indicated that Pb particles adhere to both external and internal surfaces of the membrane. Pb ions also have been found to disturb the lamellar organization by causing considerable molecular disorder within lipid layers.

Recently, it has been shown that osmotic shock, oxidative stress, and/or energy depletion activate Ca^{2+}-sensitive erythrocyte scramblase, leading to the exposure of phosphotidylserine at the cell surface. This exposure of phosphotidylserine had been implicated in the phagocytosis of RBC by macrophages that can be measured by annexin binding, as determined by fluorescence activated cell sorting analysis. Kempe et al. (2005) carried out experiments to investigate whether anemic conditions reported in Pb intoxication are the result of the decreased life span of RBCs due to the above mentioned mechanisms. These authors reported that when human RBCs were exposed to Pb-nitrate (above 0.3 μM), it caused a significant increase in Pb annexin binding, indicative of phosphotidylserine exposure. Using inhibitors for Ca^{2+}- sensitive potassium channels and whole cell patch clamp experiments, these authors concluded that Pb exposure increased activation of potassium channels, leading to shrinkage of cells and also activation of scramblase, resulting in the exposure of phosphotidylserine on the cell membrane surface. These authors further postulated that this exposure of phosphotidylserine on the membrane might have led to them being engulfed by macrophages and the ultimately decreased life span of RBCs in Pb intoxication.

Membrane Proteins

Earlier studies by Fukumoto et al. (1983) reported the differential profile for RBC-membrane polypeptides determined by SDS-PAGE analysis. These investigators found decreased levels of polypeptides in band 3 and increases in the levels of four other bands (i.e., bands 2, 4, 6, and 7) in the RBCs of human workers exposed to Pb. From these observations, they postulated that such Pb-induced alteration in RBC membrane proteins may lead to membrane permeability changes. Apostoli et al. (1988) also observed similar changes in RBC membrane polypeptides in Pb-exposed workers and suggested that band 3 may represent an anion channel protein; they also found that these changes occurred at blood Pb levels of >50 μg/100 ml.

Lead exposure has been known to increase the amount of membrane-bound protein kinase C in rat brain, endothelial, and glial cells. Belloni-Olivi et al. (1996) reported an increased phosphorylation of RBC membrane proteins on Pb exposure. When human RBCs were incubated with Pb-acetate (>100 nM) for 60 min, it was found to increase phosphorylation of membrane cytoskeletal proteins (120, 80, 52 and 45 kDa). This increase was accompanied by

increase in protein kinase C activity. Membrane proteins were not phosphorylated when treated with protein kinase C inhibitors. Calcium and diacylglycerol were found not to be involved in this process. The authors suggested that this activation of protein kinase was a direct interaction of the enzyme protein with Pb. Slobozhanina et al. (2005) reported that incubation of human RBCs with Pb-acetate (1 to 10 μM for 3 h) caused differential binding of fluorescent probes to the membrane, suggesting alterations in the physicochemical state of the membrane proteins and lipids. Based on these observations, the authors postulated that such alterations in membrane molecular composition may influence the activity of membrane enzymes and function of receptors and channels present on the membrane. These and other related studies are summarized in Annex Table AX5-2.1.

5.2.3 Effect of Lead on Erythrocyte Heme Metabolism

Enzyme studies of the heme pathway have shown that Pb is an inhibitor of several enzymes involved in heme synthesis, including 5-aminolevulinic acid dehydratase (ALAD), coproporphyrinogen oxidase, and ferro chelatase (see Figure 5-2.1 for a schematic representation of heme biosynthesis). ALAD is a cytoplasmic enzyme that catalyzes the second, rate-limiting step of the heme biosynthesis pathway; that is, ALAD catalyzes formation of porphobilinogen through the conjugation of two molecules of δ-aminolevulinic acid. ALAD is a Zn-dependent enzyme, and thiol groups are essential for its activity (Bernard and Lauwerys, 1987). Decreased erythrocyte ALAD is the most sensitive indicator of human Pb exposure, to the extent that measurement of ALAD activity reflects well Pb levels in the blood. Similarly, erythrocyte ALAD activity measurements have been used to assess Pb toxicity in other species.

Erythrocyte ALAD

Terayama et al. (1986) reported decreased ALAD activity in rat RBCs at blood Pb levels of 100 μg/100 mL. Scheuhammer (1987) studied the usefulness of the ALAD ratio (activated/nonactivated enzyme activity) to study Pb effects in avian RBCs. The ALAD activity ratio is a sensitive, dose responsive measure of Pb exposure regardless of the mode of administration of Pb. For example, dietary Pb concentrations as low as 5 ppm (dry weight) can be estimated through the use of the ALAD enzyme activity ratio method. A highly significant

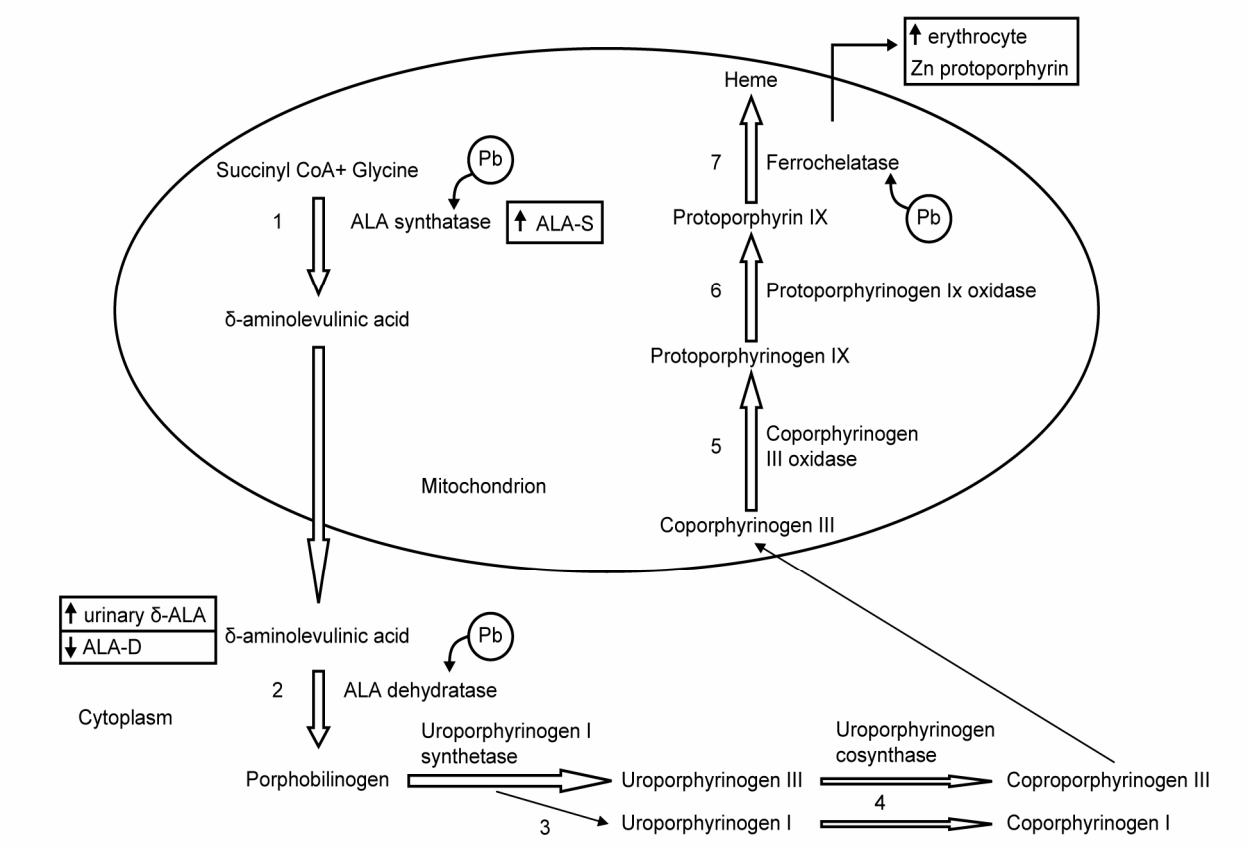

Figure 5-2.1. Schematic presentation of heme synthesis pathway. Potential lead (Pb) interacting sites are indicated by curved arrows (↑ increased, ↓ decreased).

positive correlation was observed between dietary Pb concentration over the 5 to 100 ppm range and the ALAD activity ratio. The author concluded that RBC ALAD ratio may be a useful method for estimating average dietary concentrations of Pb over an environmentally relevant range, in situations where diet is the major source of exposure to Pb or where accurate estimations of dietary Pb are not possible. Redig et al. (1991) reported heme synthetic pathway alterations upon chronic exposure (3 or 11 weeks) to Pb in red-tailed hawks. This treatment resulted in a severe decrease in RBC ALAD activity, which did not return to normal levels until 5 weeks after termination of Pb treatment. Lead exposure also decreased ALAD activity in the bone marrow and in the liver but did not alter aminolevulinic acid synthase activity. Dorward and Yagminas (1994), using comparative enzyme kinetic analysis of ALAD in Pb-exposed female cynomolgus monkeys and human erythrocyte ALAD, found similar inhibition profiles and concluded that ALAD could be a useful model for measuring the biological response in

monkeys. Santos et al. (1999) reported that rat RBC heme biosynthesis was affected by either Pb treatment alone or Pb in combination with ethanol, due to the inhibition of ALAD activity.

Analysis of blood ALAD activity had been used as a powerful clinical biomarker in evaluating Pb toxicity in occupational exposure. Fontanellas et al. (2002) further suggested that this enzyme assay be used in identifying even subclinical Pb poisoning in chronic renal failure (see Section 5.7 for details).

Other Heme Metabolism Enzymes

Taketani et al. (1985) studied the heme synthesizing activity of ferric ion using purified ferrochelatase from rat liver mitochondria and reported that Pb reduced NAD(P)H-dependent heme synthesis by 50% at 10^{-5} M, but that it had no effect when ferrous ion was used as the substrate. Based on these results, the authors concluded that heme synthesis from ferric ion was more susceptible to Pb than the ferrous ion. These studies also revealed that the NAD(P)H oxidizing system reduces ferric ion to ferrous ion, which in turn was used for heme synthesis by ferrochelatase.

The effect of various metals, including Pb, on RBC porphobilinogen synthase (PBG-S) was studied using human RBC hemolysate. Farant and Wigfield (1987) reported that the effect on the enzyme depends on the affinity of the metal for thiol groups at its active sites. Additional studies carried out by the same group utilizing rabbit erythrocyte PBG-S indicated that Pb acts as a potent effector of this enzyme both in vitro and in vivo (Farant and Wigfield, 1990). Human RBC porphobilinogen synthetase activity was found to be inhibited by Pb, while Zn ions activated this enzyme (Simons, 1995). Another enzyme involved in the heme synthetic pathway, porphobilinogen deaminase, was inhibited in human RBC by Pb-nitrate (100 mM) in in vitro studies, but had no effect in vivo (Tomokuni and Ichiba, 1990). Rossi et al. (1992) reported no inhibition of coproporphyrinogen oxidase activity in human lymphocytes on exposure to Pb. Heme synthesis can also be affected in Pb intoxication by interference with Fe transport into reticulocytes. Using a rabbit reticulocyte model, Qian and Morgan (1990) reported that inhibitory effects of Pb on transferrin endocytosis and iron transport across the membrane may also contribute to altered heme metabolism in RBCs. These and other related studies are summarized in Annex Tables AX5-2.2 and 5-2.3.

5.2.4 Effect of Lead on Other Hematological Parameters

The RBC pyrimidine 5-nucleotidase (P5N) catalysis of the hydrolytic dephosphorylation of pyrimidine 5-monophosphates is sensitive to inhibition by Pb. Tomokuni et al. (1989) evaluated the activity of RBC and bone marrow 5-nucleotidase (P5N) and RBC ALAD in mice exposed to drinking water Pb (200 to 500 ppm) for 14 or 30 days. These authors reported that Pb exposure decreased both P5N and ALAD activities in erythrocytes. Additional studies from this group, using a similar exposure regimen, indicated no change in levels of urinary coporphyrins.

Lead exposure (4 mg/kg and 6 mg/Kg body wt/30 days) in splenectomized rats was found to cause depletion of RBC Hb content, to increase numbers of reticulocytes in peripheral blood, and to increase urinary delta aminolevulinic acid excretion (Gautam and Chowdhury, 1987). These authors further reported that the increased number of reticulocytes found in the blood may be due to induced acceleration of the erythropoeitic cell series. Redig et al. (1991) reported biphasic effects of Pb on hematological parameters from their chronic exposure studies in red-tailed hawks over 3 or 11 weeks. These authors observed a rapid and relatively brief increase in RBC free protoporphyrin and a slower, but more prolonged, increase in its Zn complex with 3-week exposure to Pb (0.82 mg/kg body wt). On the other hand, exposure to a higher dose of Pb (1.64 mg/kg body wt) for a longer duration (11 weeks) resulted in a decrease in the Hct and Hb. Panemangalore and Bebe (1996) reported that Zn deficiency increased the Pb-induced accumulation of porphyrin in RBCs to a lesser extent compared to its accumulation in the liver in weaning rats.

The effects of Pb on RBC number and other Hct parameters appear to be dose dependent. Iavicoli et al. (2003) investigated these effects by feeding mice with eight different doses of Pb below (0.6 to <2.0 μg/dL) and above (>2.0 to13 μg/dL) normal background levels. These authors reported that mice receiving below normal background levels of dietary Pb displayed enhanced RBC counts and increased Hb and Hct values, whereas a marked decrease in RBC number occurred when blood Pb levels approached 10 μg/dL. Sivaprasad et al. (2003) also reported significant reductions in RBC Hb content and Hct on Pb exposure (0.02% Pb-acetate in drinking water for 5 weeks). Toplan et al. (2004) observed significant decreases in RBC Hb content and Hct and increases in blood viscosity in Wistar rats after 5-week exposure to Pb. Studies cited above are summarized in Annex Table AX5-2.4.

5.2.5 Effects of Lead on Erythrocyte Enzymes

The toxic effects of Pb on RBCs result from its complexation with the sulfhydryl, carboxyl, and imidazole groups of proteins, particularly enzymes, by competitive binding of Pb^{2+} with Zn^{2+} or Mg^{2+} in metalloenzymes. This binding of Pb to enzyme proteins can inhibit enzymes involved in the glycolytic and pentose phosphate pathway, both of which are sources of energy compounds and intermediates of purine conversion, thus causing a disruption of energy metabolism. Along with these changes, Pb-induced changes in the membrane integrity, as discussed earlier (Section 5.2.1), may also affect the enzymes' associated ion channels and other transport mechanisms.

Energy Metabolism

Erythrocytes generate high-energy ATP by anerobic glycolysis and cycle oxidized and reduced nicotinamide adenine nucleotide phosphate (NADP) by the aerobic pentose phosphate pathway. Anemic conditions associated with Pb poisoning, along with the inhibitory effects of Pb on heme synthesis, may result in increased RBC destruction due to the inhibitory effects of Pb on the activities of the enzyme, pyramidine 5-nucleotidase (P5N). Deficiency of this enzyme is characterized by intracellular accumulation of pyramidine-containing nucleotides, leading to hemolysis. Inhibition of this enzyme along with the perturbations in heme metabolism create imbalances in the energy currency of the erythrocyte. Perturbations in energy metabolism can be followed by changes in the concentration of purine nucleotides. In erythrocytes, these compounds cannot be synthesized de novo, they can only be reconstructed from preexisting free purine bases on nucleosides through salvage type reactions. The cell energy content can be measured by adenylate (ATP + ADP + AMP) and guanylate (GTP + GDP + GMP) nucleotides, and by their sum total. The concentrations of nucleoside monophosphates increase in cases of cell energy deficit, but they quickly degrade to nucleosides and bases.

Cook et al. (1987) compared P5N and deoxypyramidine-5-nucleotidase levels in the RBC of Pb-exposed workers and matched controls and reported significantly lower levels of P5N in Pb-exposed workers. Konantakieti et al. (1986) reported similar observations in neonatal rat RBCs. These authors further indicated that the low levels of nucleotides were due to inhibition of P5N activity by Pb, as the depression in enzyme activity was correlated with blood Pb levels. This was further validated by in vitro inhibition of P5N in a dose-dependent manner. Tomokuni

and Ichiba (1987) found similar results with human RBCs both in vitro and in vivo. They reported activation of Pb-exposed human RBCs. Antonowicz et al. (1990) observed significantly higher levels of glycolytic enzymes and increased production of lactic acid and 2,3-diphospho glycerol, when human RBCs were incubated with Pb. Based on their observations, these authors suggested that Pb exposure may result in anaerobic glycolysis activation in human RBCs. In contrast, Grabowska and Guminska (1996) reported that Pb exposure diminished the ATP levels in human RBCs by inhibiting aerobic glycolysis.

Erythrocyte energy metabolism in workers exposed to heavy metals, but without clinical manifestations of toxicity, was found to intensify and become more pronounced when they were occupationally exposed to Pb. Nikolova and Kavaldzhieva (1991) measured the exposed workers and reported higher ratios of ATP/ADP in Pb-exposed workers. Because the RBC energy pool is perturbed due to Pb exposure, Morita et al. (1997) evaluated the effect of Pb on NAD synthetase and reported an apparent dose-dependent decrease in NAD synthetase activity in the erythrocytes of Pb exposed workers.

Baranowska-Bosiacka and Hlynczak (2003) evaluated Pb effects on distribution profiles of adenine, guanine nucleotide pools and their degradation products in human umbilical cord RBCs. In vitro exposure equivalent (Pb-acetate; 100 to 200 μg/dL) to Pb exposure for 20 h were found to significantly lower the levels of nucleotide pools, including NAD and NADP, accompanied by a significant increase in purine degradation products (adenosine, guanosine, inosine, and hypoxanthine). Associated morphological RBC alterations were also observed, with marked significant increases in stomatocytes, spherocytes, and echinocytes. These investigators also observed similar alterations in the nucleotide pools in Wistar rat RBCs with short-term exposure to Pb (Baranowska-Bosiacka and Hlynczak, 2004). Based on these observations, the authors postulated that decreases in NAD and NADP concentrations in RBCs may be a good indicator of Pb-induced disturbance in the energy process and can serve as a useful marker for chronic Pb exposure. If NAD synthetase activity had been measured in these studies, it might have provided experimental support for the observation of inhibition of NAD synthetase reported by Morita et al. (1997).

Other Enzymes

Lead-induced efflux of K^+ from human RBCs had been recognized as being due to the ability of Pb to selectively increase the membrane permeability for this cation. Studying the efflux of ^{86}Rb using inside-out RBC vesicles, Alvarez et al. (1986) demonstrated that Pb promoted the selective efflux of K^+ ions by altering the sensitivity of Ca^{2+} binding site on the membrane either by direct binding or by altering Mg^{2+}-mediated modulation. Fehlau et al. (1989) indicated that this modulation of the Ca^{2+}-activated K^+ channel in human RBCs coincides with the activation of RBC membrane-bound oxidoreductase. These authors suggested that, even though these two are independent events, the oxidoreductase enzyme activity may influence K channel gating.

Earlier studies by Mas-Oliva (1989) on the potential effects of Pb on the RBC membrane (using RBC ghosts) indicated that Pb has inhibitory effects on Ca^{2+}-Mg^{2+}-ATPase. Further investigations on the role of calmodulin in the inhibition of Ca^{2+}-Mg^{2+}-ATPase indicated that the inhibitory activity on the enzyme may be due either to the effect of Pb on sulfhydryl groups on the enzyme or by direct binding to calmodulin.

Jehan and Motlag (1995) reported that when albino rats were administered Pb i.p (5 or 20 mg/kg body wt) for 14 consecutive days either alone or in combination with Cu (2 mg/kg body wt) or zinc (5 mg/kg body wt), there were severe decreases in RBC membrane enzyme, acetylcholine esterase (AchE), NADH dehydrogenase, and Na^+-K^+ ATPase levels along with decreases in phospholipid content, hexose, and hexosamine. Of the combined metal treatment exposure regimens, Zn was found to considerably reduce such changes. Grabowska and Guminska (1996) assayed three ATPase activities (i.e., Na^+-K^+ ATPase, Mg^{2+}-ATPase, and Ca^{2+}-ATPase) in human RBC in vitro and reported RBC Na^+-K^+ ATPase to be the only enzyme inhibited by Pb, while Ca^{2+} or Mg^{2+}ATPases were not sensitive to Pb. On the other hand, Sivaprasad et al. (2003) observed Pb-induced reductions in RBC activities of the three of those ATPase activities.

Two reports by Calderon-Salinas et al. (1999a,b) indicated Pb effects on calcium transport in human RBC. Initial studies by this group indicated that Pb and Ca are capable of inhibiting the passive transport of other metals in a noncompetitive way. Inhibition studies using N-ethyl-maleimide indicated that Pb and Ca share the same permeability pathway in human RBCs and that this transport system is electrogenic (Calderon-Salinas et al., 1999a). Additional studies by

the same group reported that Pb is capable of inhibiting Ca efflux by inhibiting Ca-ATPase (Calderon-Salinas et al., 1999b). These authors further suggested that under physiological conditions, Pb, via Ca^{2+}-ATPase, alters Ca influx, while chronic Pb intoxication inhibits Ca efflux by altering RBC calcium homeostasis. Silkin et al. (2001) reported Pb-induced activation of K channels in the RBCs of the teleost fish *S. porcus*. Exposure of teleost fish RBCs to 1 to 2 μM Pb led to a minor loss in cellular K^+; but, at 20 to 50 μM Pb, about 70% of cellular K^+ was lost. Based on their observations of Pb-induced K^+ efflux from RBCs under competitive and inhibitory regimens, these authors suggested that Pb activates RBC K^+ channels.

Eder et al. (1990) and Loipfuhrer et al. (1993) investigated activity levels of Ca^{2+}-ATPase and calcium accumulation, respectively, in Pb-depleted rat RBCs. No alteration in Ca^{2+}-ATPase activity or Ca accumulation was observed in the P0 generation (Eder et al., 1990). On the other hand, significant reduction in Ca-ATPase activity was observed in the F1 generation. It was suggested that Pb-induced alterations in the metabolism of phospho- and glycoproteins result from Pb depletion and may be responsible for the reduced enzyme activity. Both of the groups postulated that the decreased MCV observed in Pb depleted rat RBCs could be due to reduced Ca^{2+}-ATPase activity in the RBCs. These and other related studies are summarized in Annex Tables AX5-2.5 and 5-2.6.

5.2.6 Erythrocyte Lipid Peroxidation and Antioxidant Defense

Although several mechanisms have been proposed to explain Pb toxicity, no mechanisms have been defined explicitly. Recent literature on Pb toxicity suggests oxidative stress as one of the important mechanisms for toxic effects of Pb in various organs. Because RBCs accumulate major amounts of Pb compared to other tissues, oxidative stress may also result in the accentuation of lipid peroxidation with concomitant inhibition of antioxidant enzymes, such as superoxide dismutase (SOD), catalase, GSH peroxidase, GSH reductase, and simultaneous increases in oxidized GSH (GSSG) and reduced GSH/GSSG ratios. Pb-induced lipid peroxidation and the mitigating effects of experimental chelation therapy are discussed with relevance to each tissue or organ within this chapter. The discussion focuses on the available literature with reference to studies on erythrocytes.

Patra and Swarup (2000) reported significant changes in RBC lipid peroxide levels and anti oxidant defense (SOD and catalase) levels in RBC hemolysates from male calves exposed to

Pb (7.5 mg/kg body wt for 28 days). These authors suggested the potential role for increased peroxide levels in Pb-induced alterations in RBCs. Mousa et al. (2002) investigated the levels of various antioxidant enzymes, thiols, lipid peroxide in erythrocytes, and total thiol status of plasma in goats exposed to Pb (Pb-acetate, 5.46 mg/kg body wt for 2 weeks). These authors reported that all the parameters referred above were significantly increased in RBCs by day 7 and receded to normal levels by day 14, while peroxides remained significantly increased even by day 14. Based on these observations, it was suggested that Pb-induced lipid peroxide generation in RBCs appears to be a continuous process and can lead to persistent oxidative stress in RBCs with chronic exposure.

Metal chelator agents have been used clinically to reduce internal Pb body burden. These agents form an insoluble complex with Pb and are excreted. Though the majority of studies on the clinical potential of various experimental agents, including certain antioxidants, have been extensively performed mainly in relation to toxicity associated with hepatic and kidney tissues, such studies have also considered their potential effects on heme metabolism and blood Pb levels (see Sections 5.7 and 5.10). In the following paragraphs, two recent representative studies in experimental animals that specifically assessed the protection conferred to erythrocytes are described.

El-Missiry (2000) investigated the protective role of the pineal hormone, melatonin, on Pb-induced suppression of heme synthesis as a consequence of reduced antioxidant status. Intramuscular injection of Pb-acetate (10 mg/kg body wt for 7 days) caused a significant reduction in heme synthesis with decreased blood Hb levels and decreased RBC and liver ALAD. Pretreatment of rats with melatonin (30 mg/kg body wt) intragastrically prevented the suppressive effects of Pb on RBC heme metabolism by conferring protection to the antioxidant capacity of the cells and also by scavenging free radicals generated by Pb intoxication.

Sivaprasad et al. (2003) studied the protective effects of dl-alpha-lipoic acid (LA, 25 mg/kg body wt) and meso-2,3-dimercaptosuccinic acid (DMSA, 25 mg/kg body wt); and they found such treatments, either alone or in combination for a week, had an effect on alterations in RBC functions induced by Pb-acetate (0.02% in drinking water for 5 weeks). These authors reported that treatment with LA or DMSA, alone or in combination, reversed Pb-induced increased LPO and reductions in Hb and Hct, along with changes in other biochemical parameters affected by Pb treatment. These authors further concluded that combined treatment

was much more potent and effective. These and other related studies are summarized in Annex Table AX5-2.7.

5.2.7 Summary

- The 1986 Pb AQCD reported that the activity of ALAD appeared to be inversely correlated to blood Pb values and was found inhibited in several tissues. Human studies reviewed in 1986 Pb AQCD also indicated that occupational exposure to Pb results in decreased RBC survival along with alterations in RBC membrane integrity and energetics.

- More recent studies reviewed in this AQCD indicate that the transport of Pb across the RBC membrane is energy-independent, carrier-mediated and that the uptake of Pb is mediated by an anion exchanger through a vanadate-sensitive pathway.

- Lead intoxication interferes with RBC survival and alters RBC mobility. Hematological parameters, such as mean corpuscular volume (MCV), mean corpuscular hemoglobin (MCH), and mean corpuscular hemoglobin concentration (MCHC), are also significantly decreased upon exposure to Pb. These changes are accompanied by decreased membrane sialic acid content.

- Morphological analyses using electron paramagnetic resonance imaging and spin labeling techniques indicate that changes occur in RBC morphology upon Pb exposure.

- Lead-induced RBC membrane lamellar organization and decreases in membrane lipid resistance to oxidation in rats appear to be mediated by perturbations in RBC membrane lipid profiles. Similarly, Pb-induced altered phosphorylation profiles of RBC membrane proteins have been reported.

- Erythrocyte ALAD activity ratio (ratio of activated/non activated enzyme activity) has been shown to be a sensitive, dose-responsive measure of Pb exposure, regardless of the mode of administration of Pb. Competitive enzyme kinetic analyses in RBCs from both human and Cynomolgus monkeys indicated similar inhibition profiles by Pb.

- Consistent observation of Pb-mediated inhibition of pyramidine 5′-nucleotidase (P5N) suggests this enzyme as a potential biomarker for Pb exposure.

- Significant reductions in levels of nucleotide pools (e.g., NAD and NADP) accompanied by significant increase in purine degradation products have been implicated in the Pb-induced altered energetics of RBCs.

- Lead-induced increased permeability for K^+ in RBCs appears to be due to the selective efflux of K^+ ions on the RBC membrane due to altered sensitivity of the Ca^{2+}-binding site on the membrane. Erythrocyte Na^+-K^+ ATPase appears to be more sensitive to Pb-induced inhibition than Ca^{2+}-Mg^{2+} ATPase.

- Chelation agents and the pineal hormone, melatonin, have been reported to confer protection against Pb-induced lipid peroxidation and increased antioxidant defense in RBCs.

The newly available (since 1986) scientific evidence presented in this section convincingly demonstrates deleterious effects of lead on erythrocyte cell morphology, function, lead uptake and alterations in certain enzymes involved in heme synthetic pathways. However, some of the interesting and important conclusions are derived mainly from in vitro studies, often using short time incubations. It would be useful to substantiate such findings further by more systematic studies employing meaningful experimental designs for in vivo evaluation of laboratory animal models.

5.3 NEUROLOGICAL/NEUROBEHAVIORAL EFFECTS OF LEAD

5.3.1 Neurotoxicological/Neurobehavioral Effects of Lead in Animals

5.3.1.1 Introduction

Since the initial description of Pb encephalopathy in the developing rat in the mid-1960s, (Pentschew and Garro, 1966), a continuing research focus has been on defining the extent of CNS involvement at subencephalopathic, environmentally relevant, levels of exposure. These efforts have primarily addressed the developing organism, consistent with the primary public health concerns for neurotoxicity from Pb during this period. While significant research advances have been made in animal studies over the last four decades, relating these findings to neurotoxicity in children has been challenging and difficult. The barriers to greater progress have primarily been due to Pb's multiple toxic mechanisms of action in brain tissue, which encompasses variable, overlapping, and, at times, opposing dose-effect relationships. One goal of this section is to bring greater clarity to the current state of knowledge.

The Pb neurotoxicity evidence available for assessment in the 1986 Lead AQCD was considerably different in character from current, newly available findings. The literature was dominated by various types of assessments of CNS biogenic amine function in exposed animals, with dopaminergic neuronal systems seeming the most sensitive to the metal and drawing the most attention. In addition, the prevailing wisdom was that the neuronal actions of Pb were best

elucidated by perturbing neurotransmitter systems with CNS agents of known mechanism of action and comparing the responses in exposed animals to those in control subjects, an approach of limited value. Only some of those studies reported blood and/or brain Pb concentrations along with the experimental findings, rendering interpretation of results across different laboratories difficult and somewhat unreliable.

As of 1986, perhaps the most reliable evidence concerned the effects of acute exposure to Pb^{2+} in vitro on voltage-sensitive Ca^{2+} channel function in the nerve cell membrane, developed to a great extent by Cooper and co-workers (Kober and Cooper, 1976; Cooper and Manalis, 1984; Suszkiw et al., 1984). Using neuromuscular endplate or synaptosomal preparations, these studies demonstrated that Pb^{2+} interfered with Ca^{2+} influx through voltage-sensitive channels. These findings significantly advanced the field, though acute exposure in vitro bore little resemblance to environmentally relevant routes and magnitudes of exposure.

In the ensuing two decades, the Pb neurotoxicity literature has reflected an increased focus on cognitive-related mechanisms and the refinement of approaches and methodologies. Exposure-induced alterations at glutamatergic synapses have become a primary substrate of attention. Synaptic plasticity models (e.g., long-term potentiation [LTP]) developed in the 1990s are used in Pb studies in laboratories around the world. Behavioral paradigms, refined to more consistently discriminate Pb effects, aided in identifying optimal testing conditions and developmental periods for exposure. The cumulative result of these advances has lead to clearer understanding of likely mechanisms underlying Pb-induced cognitive impairments found in Pb-exposed children.

The Pb neurotoxicity evidence reviewed in this section is organized largely according to scientific discipline: neurochemical alterations involving glutamatergic, cholinergic, and dopaminergic function; mechanisms defined by neurophysiological approaches; changes in auditory and visual function; modifications in behavioral function; induced alterations in cellular morphology; and findings on cellular disposition of Pb. This type of organization permits a more focused analysis of a very extensive, broad literature.

5.3.1.2 Neurochemical Alterations Resulting from Lead Exposure

The following areas of investigation have drawn the most attention in the Pb neurotoxicity field over the last 20 years. A summary of the key studies evaluating neurochemical alterations resulting from Pb exposure are listed in Table AX5-3.1.

Lead and Neurotransmitter Release Processes

By the mid-1980s, it was evident that acute exposure to Pb^{2+} in vitro reduced the magnitude of depolarization-induced transmitter release, apparently by inhibiting Ca^{2+} influx into the nerve ending through voltage-sensitive Ca^{2+} channels (Kober and Cooper, 1976; Cooper and Manalis, 1984; Suszkiw et al., 1984). Since then, several investigators utilizing various preparations (Shao and Suszkiw, 1991 [cortical synaptosomes]; Tomsig and Suszkiw, 1993 [bovine chromaffin cells]; Braga et al., 1999a,b [cultured hippocampal cells]; Westerink and Vijverberg, 2002 [PC12 cells]), have demonstrated that, in the absence of Ca^{2+}, Pb^{2+} exhibits Ca^{2+}-mimetic properties in stimulating exocytosis and is substantially more potent in doing so. That is, in the absence of Ca^{2+} and depolarization, nM concentrations of Pb^{2+} alone stimulate transmitter release. Many investigators have proposed that this action, in conjunction with the ability of Pb^{2+} to suppress evoked release, produces a higher noise level in synaptic transmission in Pb-exposed animals.

The ability of Pb to diminish stimulated transmitter release has been demonstrated in intact chronically exposed animals via the use of intracerebral microdialysis (Kala and Jadhav, 1995; Lasley and Gilbert, 1996; Lasley et al., 1999). More recently, Lasley and Gilbert (2002) used Ca^{2+}-free perfusate containing a Ca^{2+} channel antagonist for microdialysis to identify the Ca^{2+}-independent component of release. These workers demonstrated that under these conditions high K^{+}-stimulated glutamate and GABA release were *elevated* in chronic high level Pb-exposed animals, suggesting a Pb^{2+}-induced enhancement of evoked release. It was concluded that this pattern of results indicated the presence of two actions of Pb on transmitter release in vivo: (1) a more potent suppression of stimulated release seen at lower exposure levels (associated with blood Pb values of 27-62 µg/dL) combined with (2) Ca^{2+}-mimetic actions that independently induce the exocytosis seen at higher exposure levels (associated with blood Pb values $\geq$62 µg/dL). Together, these two actions produce a biphasic dose-effect relationship (see

Figure 5-3.1). Thus, there is good correspondence between findings in in vitro and in vivo studies with respect to the actions of Pb on transmitter release.

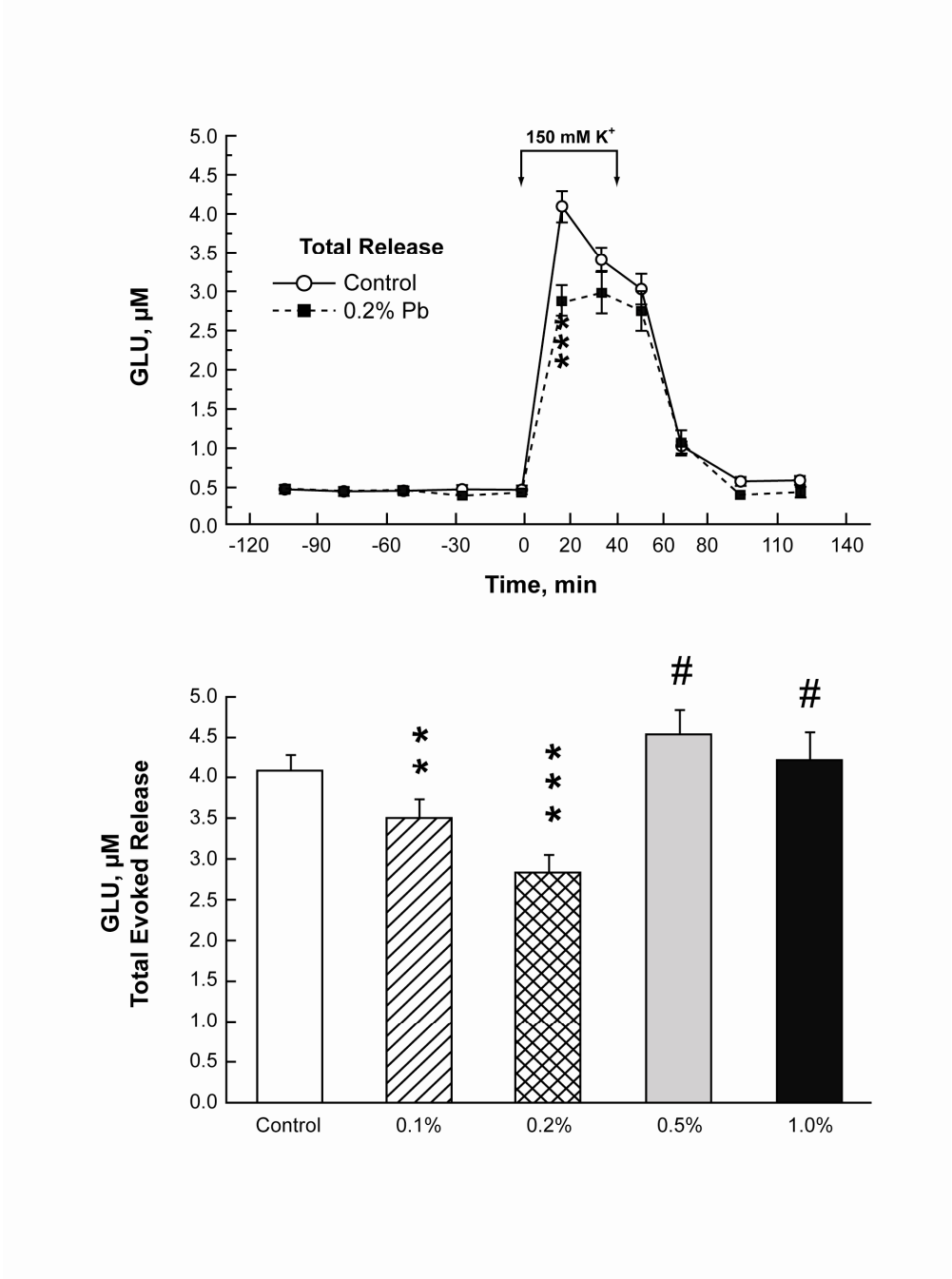

Figure 5-3.1. Time course of extracellular GLU concentration and GLU concentration in response to lead exposure.

*** p<0.001; ** p<0.01 relative to the GLU concentration in control animals; #p<0.0001 relative to the GLU concentration in the 0.2% Pb group.

Source: Lasley and Gilbert (2002).

Lead and Glutamatergic NMDA Receptors

Because of the established importance of the NMDA subtype of glutamate receptor in synaptic plasticity and learning, these receptors have been a focus of intense interest in Pb neurotoxicity for the last 15 years. Using whole cell and single channel patch clamp methodologies, Alkondon et al. (1990) were the first to report that Pb^{2+} inhibited the function of the NMDA receptor channel complex. Guilarte and Miceli (1992) reported similar findings using nominal Pb^{2+} concentrations and receptor binding techniques, and drew parallels between Zn^{2+} Ca^{2+2}- and Pb^{2+}-induced inhibition of the channel. However, Lasley and Gilbert (1999), using free Pb^{2+} ion concentrations and radioligand binding, demonstrated that, despite the similarities, Pb^{2+} did not inhibit the NMDA receptor channel complex by binding to the Zn^{2+} allosteric site. Furthermore, they indicated that the Pb^{2+} IC_{50} of 0.55 μM for inhibition of the channel complex was likely approximately two orders of magnitude greater than the extracellular fluid concentrations of Pb^{2+} associated with environmentally relevant exposure. This does not mean that NMDA receptor function does not change after Pb exposure, but it strongly suggests that the alterations are not based on a direct Pb^{2+} action.

Unfortunately, a consensus on the effects of chronic Pb exposure on NMDA receptor expression and function has not been achieved. Extensive effort has been invested to assess NMDA receptor subunit mRNA and protein expression in exposed animals (Guilarte and McGlothan, 1998; Nihei and Guilarte, 1999; Guilarte et al., 2000; Nihei et al., 2000; Toscano et al., 2002; Guilarte and McGlothan, 2003), but consistent findings have not emerged. An exception was perhaps the work of Nihei et al. (2000) who found decreases in hippocampal NR1 subunit mRNA and protein expression deficits in LTP to be associated with impaired spatial learning in PB-exposed animals. Correlations of this type with functional measures are valuable in validating the biochemical observations.

While exposure-induced alterations of NMDA receptor binding have been observed in multiple laboratories, there has not been uniform agreement as to the direction of change. Upregulation of NMDA receptor density has been observed in rats continuously exposed throughout development (Ma et al., 1997; Lasley et al., 2001), but receptor downregulation has also been reported when exposure was begun immediately postweaning (Cory-Slechta et al., 1997a). The results of behavioral investigations are best explained by increases in NMDA receptor density. Cohn and Cory-Slechta (1993, 1994b), using a repeated learning component of

a multiple reinforcement schedule, observed enhanced performance sensitivity to exogenous NMDA administration and diminished sensitivity to MK-801, an NMDA receptor antagonist in exposed animals. The same findings resulted when the drug discrimination paradigm was utilized (Cory-Slechta, 1995a; Cory-Slechta et al., 1996b): enhanced sensitivity to NMDA and reduced sensitivity to MK-801 in Pb-exposed groups. A decreased sensitivity to MK-801 can result from either increased numbers of NMDA receptors or a diminished access of the antagonist to its binding site in the ion channel. Thus, all these behavioral observations may be accounted for by Pb-induced increases in NMDA receptor density resulting in increased sensitivity to agonists coupled with decreased sensitivity to antagonists. That is, the functional measures suggest that an NMDA receptor upregulation occurs.

Pb^{2+} and Protein Kinase C

Another important focus area for Pb neurotoxicity research has been the interactions of Pb^{2+} with protein kinase C (PKC) activity. Markovac and Goldstein (1988a) were the first to report that Pb^{2+} directly stimulated PKC activity at picomolar concentrations, thereby exhibiting greater potency for this action than Ca^{2+} by 4-5 orders of magnitude. Long et al. (1994) made similar observations using free Pb^{2+} and Ca^{2+} ion concentrations and nuclear magnetic resonance spectroscopy, resulting in an EC_{50} of 55 pM for Pb^{2+} stimulation of PKC. These workers also presented evidence suggesting that the maximal efficacy of Pb^{2+} was less than that of Ca^{2+}, despite its greater potency. Tomsig and Suszkiw (1995) elegantly elucidated multiple interactions of Pb^{2+} with PKC, identifying both stimulatory (affinity in the pM range) and inhibitory (affinities in the nM and μM range) binding sites on the kinase. They also showed that, on the basis of these interactions, Pb^{2+} induced a peak efficacy for stimulation of PKC that was only ~40% of the maximal efficacy produced by Ca^{2+}, leading to their terming Pb^{2+} a partial agonist of the kinase, as reflected in Figure 5-3.2.

Subsequent studies have begun to examine the cellular impact of the Pb^{2+} effects on PKC. Kim et al. (2000) showed that acute Pb^{2+} exposure in vitro stimulated immediate early gene expression in cultured cells by a mechanism that requires PKC. Braga et al. (2004) have demonstrated that Pb^{2+} stimulation of PKC results in inhibition of nicotinic cholinergic modulation of glutamate and GABA synaptic transmission in cultured hippocampal cells. It is anticipated that future studies will further develop this line of investigation.

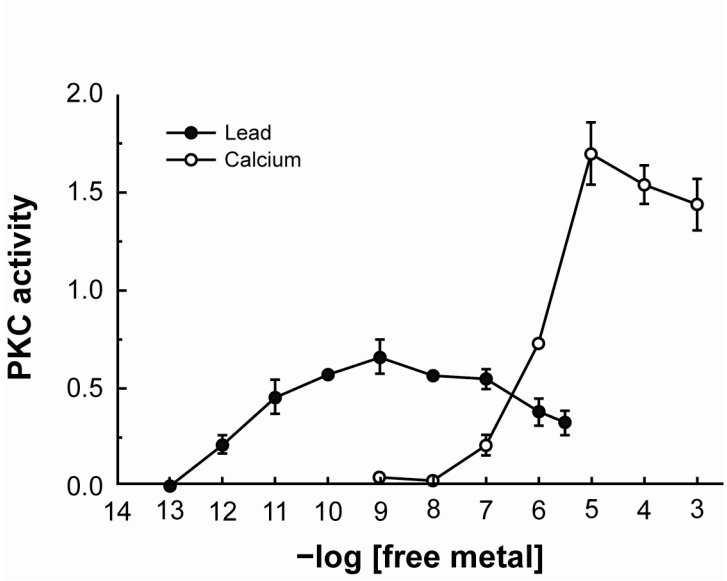

Figure 5-3.2. PKC activity as a function of Ca^{2+} and Pb^{2+} concentrations.

Source: Tomsig and Suszkiw (1995).

The effects of chronic Pb exposure on PKC signaling have been more difficult to discriminate. Most investigators have utilized broken cell preparations and measures of either kinase translocation or enzyme activity; however, a broken cell preparation has not been shown to simulate the intracellular milieu of a chronically exposed intact animal. In the preparation of a tissue extract for determination of kinase activity, the unbound Pb^{2+} is removed or greatly diluted, so that the resulting activity measure largely reflects changes in total PKC expression resulting from the exposure. That is, this measure does not identify a synaptic pool of PKC or necessarily represent the pool of kinase involved in signal transduction. Alternatively, the translocation of kinase from a cytosolic to membrane cellular fraction is a somewhat nonspecific measure and observed changes should be independently confirmed. From the effects of acute Pb^{2+} exposure in vitro it is abundantly clear that PKC is a toxicologically significant intracellular target for Pb^{2+}. However, various investigators have been unable to define how this acute effect translates, if at all, to chronic exposure in the intact animal. Neither is it evident how one could discriminate inhibition of PKC activity (e.g., resulting from decreased efficacy relative to that associated with Ca^{2+}) from downregulation of the enzyme from prolonged stimulation. Resolution of these issues awaits the development of more specific methodologies.

Pb^{2+}-Ca^{2+} Interactions

In general Pb^{2+}-Ca^{2+} interactions have long been proposed as important factors in manifestations of cellular Pb toxicity and have been under investigation since before the 1986 AQCD was prepared. The classical effects of Pb^{2+} mentioned earlier include inhibiting Ca^{2+} influx through cell membrane voltage-sensitive Ca^{2+} channels and exhibiting Ca^{2+}-mimetic properties at multiple intracellular proteins. In addition, Pb^{2+} is known to disturb intracellular Ca^{2+} homeostasis (Simons, 1993b). Ca^{2+}-dependent proteins whose actions have been reported to be stimulated by Pb^{2+} include calmodulin and calmodulin-dependent phosphodiesterase (Goldstein, 1993), calcineurin (Kern and Audesirk, 2000), and Ca^{2+}-ATPase (Ferguson et al., 2000). These actions of Pb^{2+} are thought to be the points of initiation of much of the metal's cellular toxicity.

Lead Exposure and Cholinergic Neuronal Systems

The actions of chronic exposure have also been studied with respect to changes in CNS cholinergic systems, as another substrate thought to underlie cognitive function. Bielarczyk et al. (1996) reported (a) decreased functional cholinergic innervation in the hippocampus and (b) depression of choline acetyltransferase activity in hippocampus and cortex in young adult rats exposed to Pb only during early development. Similar changes were reported by Bourjeily and Suszkiw (1997), leading to the conclusion that perinatal exposure results in a loss of septohippocampal cholinergic projection neurons that persists until testing in young adulthood. Tian et al. (2000) exposed PC12 cells to Pb^{2+} for $\leq$48 h and found that the downregulation of choline acetyltransferase activity reflected the effects of the metal at the level of gene expression. Consistent with these other findings, Jett et al. (2002) employed a similar perinatal exposure protocol and observed increased nicotinic receptor binding in multiple brain regions. These reports reinforce the belief that Pb exposure during early development deleteriously affects cholinergic function and indicate that these actions are an important component of the cognitive impairment resulting from exposure to the metal.

Summary

In reviewing the Pb neurotoxicity literature of the last 20 years and the research focus areas presented above, it is evident that the effects of Pb exposure on components of

neurotransmitter release and Pb^{2+}-Ca^{2+} interactions are closely intertwined. Exposure-induced decreases in glutamatergic, cholinergic, and dopaminergic transmission are most prominent because of the purported role of these neuronal systems in brain development and cognitive function. In contrast, the weight of the data suggest an upregulation of NMDA receptors resulting from chronic exposure, but a consensus on the effects of Pb on expression and function remains to be attained, and it is increasingly apparent that this glutamate receptor subtype may not be a primary target of chronic exposure in the intact animal. While the in vitro interactions of Pb^{2+} and PKC have been carefully described and are broadly relevant to cellular signaling pathways, meaningful and valid observations of the functional effects of these interactions in intact animals have not been achieved.

5.3.1.3 Actions of Lead Exposure Defined by Neurophysiological Approaches

One of the most significant advances in Pb neurotoxicity research over the last two decades is the widespread application of synaptic plasticity models to studies of the effects of exposure. Key studies are listed in Table AX5-3.2. The incorporation of these paradigms into Pb studies could be seen as a natural progression, and one might expect that they would receive greater use in neurotoxicology, as they have in the broader field of neuroscience.

Chronic Lead and Models of Synaptic Plasticity

Throughout the 1990s, the LTP model of synaptic plasticity was utilized in studies of Pb neurotoxicity in laboratories around the world, undoubtedly because it was widely accepted that the model invoked synaptic processes that also were involved in learning and cognitive function. These investigations resulted in large body of evidence that characterized the actions of chronic exposure across several experimental parameters (see Table 5-3.1). Furthermore, at least in the hippocampal subregions, CA1 and dentate gyrus, there was uniform agreement as to the alterations that resulted.

Chronic developmental Pb exposure decreased the magnitude of LTP and increased the threshold for induction (Altmann et al., 1993; Gilbert et al., 1996; Gutowski et al., 1998; Ruan et al., 1998). Simultaneous assessments of paired-pulse functions also uncovered reductions in paired-pulse facilitation, indicative of reduced glutamate release (Lasley and Gilbert, 1996;

Table 5-3.1. Chronic Lead Exposure and LTP

Recording Site	Exposure Period[1]	Blood Pb[2]	Brain Pb[3]	Preparation	Effect of Exposure on LTP
Hippocampal Dentate Gyrus					
Gilbert et al. (1996)	P0 – P90-120	37.2	ND	in vivo	elevated induction threshold
Ruan et al. (1998)	P0 – P90-115	30.1	180	in vivo	diminished magnitude
Gilbert et al. (1999a)	G16 – P130-210 P30 – P130-210	40.2 38.7	378 350	in vivo	elevated induction threshold and diminished magnitude
Gilbert et al. (1999b)	G16 – P120-180	26.8[4] 40.2 61.8	220 378 670	in vivo	elevated induction threshold and diminished magnitude
Gilbert and Mack (1998)	G16 – P210-540	ND	ND	in vivo	accelerated decay
Hippocampal CA1					
Altmann et al. (1993)	G0 – P70-210	14.3	160	slices	blocked, required exposure during early development
Gutowski et al. (1998)	G0 – P90-130	16.0	135	slices	diminished magnitude
Hippocampal CA3					
Gutowski et al. (1997)	G0 – P13-140	28.5	180	slices	no effect across 4 ages
Gutowski et al. (1998)	G0 – P90-130	16.0	135	slices	no effect

[1]Exposure duration in terms of gestational (G) or postnatal (P) days of age; P0 = day of birth.
[2]Values expressed as µg/100 ml.
[3]Values expressed as ng/g tissue.
[4]Different blood Pb values generated by differing levels of exposure.

Ruan et al., 1998). It was also shown that the potentiation produced in Pb-exposed animals decayed more rapidly than in controls (Gilbert and Mack, 1998).

Gilbert et al. (1999a) compared the effects on LTP when exposure occurred during different developmental periods. These workers found that animals whose exposure began shortly after weaning exhibited the same impairments in LTP as animals continuously exposed from late gestation when testing in both groups occurred well into adulthood. A smaller effect on potentiation was observed when exposure was restricted to the late gestation/weaning period.

Gilbert et al. (1999b) also examined the effects of Pb on LTP as a function of chronic exposure level, utilizing a range of 0.1–1.0% Pb in the drinking water (corresponding to blood Pb values of 26 to 117 μg/dL). A reduced capacity for LTP was found at all exposure levels except in the 1.0% groups, indicative of a biphasisc dose-effect relationship (Figure 5-3.3). The 1.0% Pb-exposure level was clearly less effective than the lower exposure groups in reducing LTP magnitude and did not differ significantly from control values. Blood Pb values were elevated as a function of increasing exposure and could not account for the lack of effect in the 1.0% exposure group.

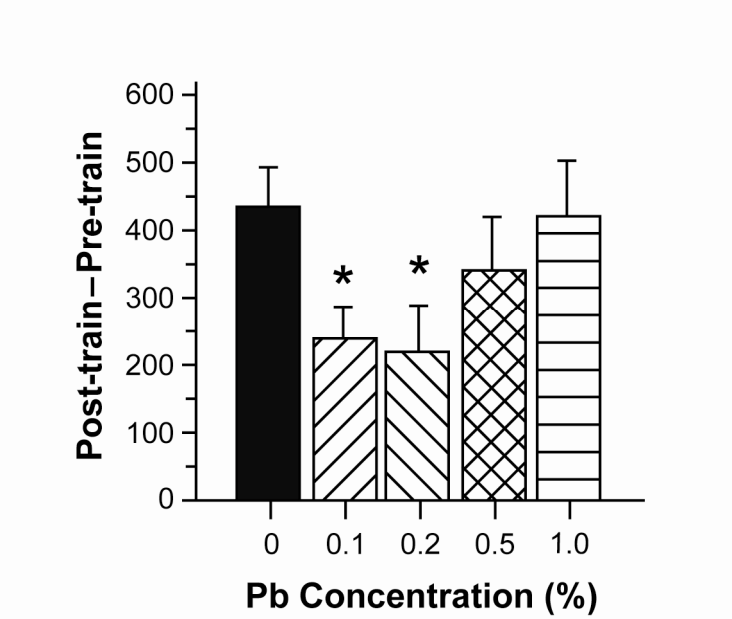

Figure 5-3.3. I/O function difference score–PS amplitude.

Source: Gilbert et al. (1999b).

Zhao et al. (1999) utilized low frequency electrical stimulation in the paradigm of long-term depression (LTD) and found that chronic Pb exposure depressed the magnitude of this form of synaptic plasticity in both hippocampal CA1 and dentate gyrus subregions. The authors also made the point that in combination with the reduced magnitude of LTP reported by other workers, the decrease in LTD magnitude results in a reduced range of synaptic plasticity in chronically exposed subjects.

While the effects of Pb on synaptic plasticity are quite similar in the CA1 and dentate gyrus regions, they are not uniformly present throughout this brain area. Gutowski et al. (1997, 1998) were unable to find any effect of chronic Pb exposure on LTP in hippocampal CA3 (i.e., mossy fiber LTP), even when the investigation was extended across multiple ages. The bases for this distinction await future investigation.

Lead Exposure, Glutamatergic Transmission, and Synaptic Plasticity

Investigation of the synaptic processes underlying LTP has provided insight into the bases for Pb exposure-related impairment of potentiation and cognitive ability (Lasley and Gilbert, 2000). Biochemical and neurophysiological approaches (Lasley and Gilbert, 1996; Gilbert et al., 1996; Ruan et al., 1998) have found stimulated glutamate release to be diminished in hippocampus at blood Pb values where deficits in LTP have been observed. Multiple actions of Pb may be involved at this exposure level because animals exposed postweaning exhibited similar decrements in evoked glutamate release to those exposed continuously from conception (Lasley et al., 1999), similar to the observations for measures of LTP. A biphasic dose-effect relationship was also found in which stimulated glutamate release in hippocampus was decreased at intermediate exposures, but not at higher levels (Lasley and Gilbert, 2002). On the basis of these observations, it is apparent that decreases in stimulated glutamate release are a significant contributing factor to the Pb exposure-related changes seen in LTP.

In comparison to the concordance across laboratories with regard to effects of chronic Pb exposure on LTP and the notable similarities to its actions on glutamate release, the effects of exposure on the NMDA receptor are relatively variable. That is, there is not widespread agreement as to the nature of the exposure-induced changes. Alterations in receptor function occur readily in response to externally applied treatments and might be expected to vary in a dynamic fashion as a function of exposure parameters (e.g., Lasley et al., 2001). However, most studies have involved measures of NMDA receptor integrity in adult animals exposed to constant levels of Pb for at least three, and more commonly 6 to 15, months, so that receptor-mediated effects should have stabilized. Consequently, the following alternative conclusions could be proposed regarding the actions of Pb exposure on the NMDA receptor that are related to its effects on LTP. First, changes in NMDA receptor function may depend on specific Pb exposure conditions. For example, a postweaning exposure protocol may not necessarily produce similar

effects to an exposure protocol initiated during earlier development. Alternatively, exposure effects on LTP may be produced at signal transduction or other cellular loci that exert regulatory influences on the NMDA receptor. This latter conclusion implies that changes in the NMDA receptor do not mediate the primary action of Pb on LTP. In addition, this indicates that identification of some site of direct Pb effect that has regulatory influence on the receptor would produce more consistently observable findings.

Lead and Electrophysiological Changes in Dopaminergic/Cholinergic Systems

Electrophysiological approaches have been employed to delineate other interesting findings in Pb-exposed animals not directly related to synaptic plasticity. Using standard extracellular recording methods, Tavakoli-Nezhad et al. (2001) identified an exposure-dependent decrease in the number of spontaneously active dopamine cells in the substantia nigra and ventral tegmental area, but they found no evidence that this decrease was related to a physical loss of cells. In subsequent work, Tavakoli-Nezhad and Pitts (2005) determined that the decrease in the number of active dopamine cells was not based on depolarization inactivation. However they were able to discern a reduced impulse flow in dopamine neurons and a diminished sensitivity of D_1 receptors in the nucleus accumbens.

The actions of Pb^{2+} on cholinergic nicotinic receptors have been investigated in acutely dissociated or cultured hippocampal cells using the patch clamp technique in whole cell mode (Ishihara et al., 1995). These workers found that Pb^{2+} potently inhibits activation of fast-desensitizing nicotinic currents in a noncompetitive and voltage-dependent manner. The nicotinic receptors affected (methyllycaconitine-sensitive) were more sensitive to Pb^{2+} than other nicotinic subtypes and are known to be highly permeable to Ca^{2+}. This latter observation likely explains the potency for their inhibition by Pb^{2+}.

5.3.1.4 Lead Exposure and Sensory Organ Function

Another focus area for Pb neurotoxicity research that has generated valuable and relevant scientific findings has been sensory organ function. Visual and auditory systems have received the most attention, have generated results closely resembling clinical observations, and have been successful in defining some of the mechanisms underlying the exposure-induced alterations. These studies are summarized in Table AX5-3.3.

Sensory Organ Assessments in Nonhuman Primates

Lilienthal and Winneke (1996) tested monkeys continually exposed to Pb from gestation through 8 to 9 years of age and found increased latencies for waves I, II, and IV in brainstem auditory evoked potentials. These effects persisted for at least 18 months after exposure was terminated and blood Pb values had declined nearly to control levels, leading to the conclusion that these actions of Pb were not dependent on current exposure. Rice (1997) determined pure tone detection thresholds in monkeys exposed continually from birth to 13 years of age, and reported that half of the subjects exhibited thresholds outside of the control range at some frequencies. These findings are consistent with reported alterations in auditory function in humans exposed to Pb developmentally (Otto and Fox, 1993).

Visual function was assessed in monkeys by Reuhl et al. (1989), who exposed a high- and a low-dose group from birth to 6 years of age. This investigation uncovered a decrease in neuronal volume density in cortical areas V1 and V2 in the high-exposure compared to the low-exposure group. These workers also found a decrease in dendritic arborization in pyramidal neurons in these brain areas, leading to the conclusion that chronic developmental Pb exposure produces changes in cytoarchitecture in visual projection areas.

Retinal Function in Rodents

The actions of Pb on retinal cells have been a focus of research for more than two decades. It has long been recognized that Pb^{2+} exhibits a selective effect on rod cells (Fox and Sillman, 1979) and, more recently, that the associated loss of rod and bipolar cells was due to exposure-induced apoptotic changes (e.g., Fox et al., [1997]). These observations have been linked with exposure-related alterations in rod-mediated visual function. In vitro studies utilizing free Pb^{2+} ion concentrations have done much to elucidate the mechanistic bases of these observations.

These latter efforts have established the concentration-dependent inhibition of cyclic GMP (cGMP) hydrolysis by free Pb^{2+}, in addition to increases in retinal cGMP and rod Ca^{2+} levels (e.g., Srivastava et al., [1995]). Kinetic studies utilizing purified rod cGMP phosphodiesterase have shown that pM Pb^{2+} concentrations competitively inhibit the enzyme relative to mM concentrations that are required for Mg^{+2} cofactor activity, thus binding with 10^4-10^6-fold higher affinity than Mg^{+2} and preventing cGMP hydrolysis (Srivastava et al., 1995).

When retinas are incubated in Ca^{2+} and/or Pb^{2+} in vitro, the rods selectively die by apoptosis associated with mitochondrial depolarization, release of mitochondrial cytochrome *c*, and increased caspase activity (He et al., 2000). He et al. (2003) have proposed that apoptosis is triggered by Ca^{2+} and Pb^{2+} overload due to translocation of cytosolic Bax to the mitochondria, which likely sensitized the overloaded mitochondria to release cytochrome *c*. Subsequent work found the elevations in free Ca^{2+} and Pb^{2+} to be localized to photoreceptors and determined that the effects of the two ions were additive and blocked by a mitochondrial permeability transition pore inhibitor (He et al., 2000). This suggested that the two ions bind to the internal metal binding site of this pore and, thereby, initiate the apoptosis cascade.

These mechanisms are entirely consistent with electroretinogram (ERG) changes observed in animals chronically exposed during early development: decreases in maximal ERG amplitude, decreases in absolute ERG sensitivity, and increases in mean ERG latency that were selective for rod photoreceptors (Fox and Farber, 1988). Also in agreement with these mechanisms are observed elevations in retinal cGMP levels and reductions in light-activated cGMP phosphodiesterase activity. Moreover, the exposure level-dependent degeneration of rod and bipolar cells exhibited the classical morphological features of apoptotic cell death (Fox et al., 1997). Other measures of visual function in chronically exposed animals also have been found to be consistent with the mechanistic data. Long-term dose-dependent elevations in response thresholds are present but only at scotopic (i.e., rod-mediated) backgrounds, and dark adaptation is delayed (Fox et al., 1994). In addition, exposure-induced decreases in rhodopsin content that were proportional to the loss of rod cells have been reported (Fox et al., 1997) as well as dose-dependent decreases in retinal Na^+, K^+-ATPase activity (Fox et al., 1991).

These studies investigating rod photoreceptors are perhaps the best examples of the ability to correlate data obtained in vitro with findings derived from in vivo exposure and with changes in visual physiology. In multiple instances, the same cellular mechanisms are affected with each approach and are consistent with ERG and rod-mediated functional measures. These relationships are summarized in Table 5-3.2.

5.3.1.5 Neurobehavioral Toxicity Resulting from Lead Exposure

The breadth of research examining Pb neurotoxicity utilizing behavioral approaches is quite diverse with respect to test paradigms, exposure parameters, test species, and

Table 5-3.2. Mechanisms of Pb-Induced Impairment of Retinal Function

In Vitro Evidence	In Vivo Evidence	Physiological Changes
Competitive inhibition of cGMP PDE	Increased retinal cGMP	
Increased retinal cGMP	Decreased stimulated cGMP PDE activity	Decreased maximal ERG amplitude Decreased absolute ERG sensitivity Increased mean ERG latency
Increased rod [Ca^{2+}]		
Apoptosis from increased photoreceptor Ca^{2+}/Pb^{2+} via binding to mitochondrial permeability transition pore	Morphological features of apoptotic rod, bipolar cell death Decreased rhodopsin proportional to cell loss Translocated cytosolic Bax to the mitochondria, cytochrome *c* released	Increased response thresholds at scotopic backgrounds Delayed dark adaptation
Decreased retinal Na^+, K^+-ATPase activity	Decreased retinal Na^+,K^+-ATPase activity	

Abbreviations: PDE, phosphodiesterase; ERG, electroretinogram.

neuropharmacological agents used. This large literature, summarized in Table AX5-3.4, has permitted development of insightful generalizations while, at the same time, providing focused descriptions of specific behaviors. In addition, the accumulated evidence has supported the development of more effective and refined methodologies.

Lead-Induced Alterations of Behavior – Nonhuman Primates

In reviewing the results of behavioral investigations of Pb neurotoxicity in nonhuman primates conducted over the last two decades, it is abundantly clear that the results are shaped by the nature of the test paradigm and the developmental exposure periods utilized. Thus, studies employing nonspatial discrimination reversal (Rice and Gilbert, 1990a), spatial delayed alternation (Rice and Gilbert, 1990b), and spatial discrimination reversal (Rice, 1990) produced observations that are distinctly different. Experimental groups continually exposed to Pb from birth to testing as adults typically exhibit learning deficits, but groups continually exposed beginning after weaning or whose exposure from birth is terminated during development may or

may not display differences from control animals depending on the sensitivity to exposure of the test paradigm.

Nonetheless, some characteristics of experimental subjects can be gleaned from investigations of neurobehavioral toxicity in nonhuman primates. Modifications of experimental parameters that make task acquisition or retention more challenging (Rice and Gilbert, 1990b; Rice, 1990) are more likely to elicit exposure-related changes in responding. In test paradigms based on fixed interval reinforcement or differential reinforcement of low rate responding schedules, Pb-exposed subjects displayed decreased inter-response times and a greater ratio of responses per earned reinforcement (Rice, 1992a,b). Exposed animals also are less sensitive to changing reinforcement contingencies and, therefore, commit more perseverative errors in responding (Rice, 1992c; Newland et al., 1994). Not surprisingly, it has been noted that these experimental behavioral effects correspond reasonably well to epidemiologic observations in Pb-exposed children (Rice, 1996; Lasley and Gilbert, 2000), thus validating the use of this species as an exposure model.

Lead-Induced Alterations of Behavior – Rodents

The observations of Pb's neurobehavioral effects in rodents in many ways resemble those conclusions attained with nonhuman primates. However, the test paradigms utilized for rats have been somewhat more refined, and the behavioral data have been subjected to more detailed analyses. As a result, valuable insights into the component mechanisms underlying the exposure-related changes have been achieved.

An olfactory serial reversal paradigm was utilized to demonstrate Pb-induced impairments in learning reversals (Hilson and Strupp, 1997; Garavan et al., 2000). These workers found that, when presented with altered reinforcement contingencies for the reversals, rats whose exposure was limited to early development exhibited a shortened initial period of responding to the previously correct cue coupled with a prolonged postperseverative learning phase for the new task. Hilson and Strupp (1997) concluded that the impaired reversal learning was due to a deficiency of learning new contingencies of the task (i.e., an associative deficit), and not based on inflexibility or deficient inhibitory control. Subsequent work by Garavan et al. (2000) determined that this associative deficit was based on a response bias and an impaired ability to associate cues and/or actions with their affective consequences.

Employing a visual discrimination task, Morgan et al. (2000) found that as the level of Pb exposure restricted to early development increased, learning of the task slowed and the number of defined "impaired" animals increased. The authors concluded that the deficits were not limited to attentional function and that an associative deficit had resulted along with a tendency to respond more rapidly. Subsequent work with visual discrimination vigilance tasks found that animals exposed only during gestation and/or lactation exhibited impaired response initiation and increased omission errors, indicating a lasting deficiency in sustained attention and an increased reactivity to errors (Morgan et al., 2001). The authors concluded that the effects of exposure are determined not only by the paradigm, but also by the timing and intensity of exposure. Cory-Slechta (2003) came to a similar conclusion on the factors underlying the manifestation of Pb effects but suggested that the alterations in attention may be due to impulsivity or aversion to delays.

The actions of early Pb exposure on memory appear to be task-dependent, but this issue has not been clearly defined. Alber and Strupp (1996) found that exposed rats performed more poorly on a series of spatial alternation tasks but that the deficit did not vary across intertrial delays, suggesting that memory was not impaired. Murphy and Regan (1999) used a one-trial, light/dark, passive avoidance paradigm and observed a decrease in recall latency on post-training day 5 in rats whose exposure was restricted to early development. Since there was no exposure effect evident during the first 48 h after training, these authors concluded that the impairment was associated with long-term memory storage. Further studies are needed to more clearly characterize the effects of chronic Pb on memory function.

Interactions of Lead Exposure and Responding to Cocaine

Behavioral responses to a number of neuropharmacological agents have resulted in important and useful insights into Pb neurotoxicity. One approach that has been unique and has produced scientifically important results has been investigation of the interactions of chronic Pb exposure and responses to cocaine. Chronic exposure of adult male rats has been shown to attenuate cocaine-induced locomotor activation (Grover et al., 1993) and result in a slower development and reduced magnitude of cocaine-induced sensitization of locomotor activity (Nation et al., 1996). The latter observations are consistent with other evidence of impaired synaptic plasticity that were presented earlier in this chapter. These actions of exposure are not

specific to cocaine, as a similar exposure regimen attenuated the reinforcing effect of brain stimulation of the medial forebrain bundle (Burkey and Nation, 1994), the nerve tract conveying nigrostriatal and mesolimbic dopaminergic neurons to forebrain regions.

Using the drug discrimination paradigm, Miller et al. (2001) restricted Pb exposure to the gestational and lactational periods of early development and observed decreased sensitivity to dopamine D_1 and D_2 receptor agonists when the animals were tested as adults. These findings may be taken as evidence of receptor downregulation, but, in this behavioral task, subjects received chronic intermittent doses of the training drug, which in this study was a low dose of cocaine. Thus, the actions of exposure on dopaminergic systems may be confounded with the receptor changes induced by chronic drug administration.

In contrast to the attenuating effects of chronic Pb administration to adults described above, exposure restricted to the gestational and lactational periods exerts potentiating effects on other types of responses to cocaine when animals are tested long after exposure is terminated. Nation et al. (2003) trained rats to self-administer cocaine intravenously, extinguished the response, and then used a systemically administered priming dose of the drug to initiate a relapse response. Exposed animals were found to have an increased sensitivity to cocaine relapse compared to identically treated controls. When multiple cocaine doses were provided, identically exposed animals were found to self-administer more of a low dose of the drug and less of a high dose than controls, again suggesting an enhanced sensitivity to the actions of cocaine (Nation et al., 2004). Finally, animals exposed to Pb in this manner were found to have an accelerated rate of acquisition of cocaine self-administration behavior (Rocha et al., 2005).

Lead Exposure and the Stimulus Properties of Neuropharmacological Agents

The drug discrimination paradigm has been utilized more widely in Pb neurotoxicity research to characterize postsynaptic receptor status for multiple neurotransmitter systems and has resulted in some useful findings. Rats chronically exposed beginning at weaning and tested as adults were trained to discriminate either a systemically administered D_1 or D_2 receptor agonist (Cory-Slechta and Widzowski, 1991). Exposed rats learned the discrimination task more rapidly than controls and exhibited greater levels of response to lower doses of the training drugs and less blockade by a D_2 receptor antagonist, consistent with generalized dopaminergic receptor supersensitivity. In groups of animals exposed only from birth to weaning and trained to

discriminate the same drugs, the D_2-D_3 subtype receptor supersensitivity in exposed animals was again present, but no changes in responding to the D_1 agonist were apparent (Cory-Slechta et al., 1992). Further work with this test paradigm employing the postweaning exposure protocol failed to demonstrate any D_1-D_2 receptor interactions in the supersensitivity displayed by Pb animals (Cory-Slechta et al., 1996a).

To test cholinergic sensitivity in animals chronically exposed after weaning, rats were trained to discriminate a muscarinic agonist (Cory-Slechta and Pokora, 1995) and were tested in the added presence of a muscarinic antagonist. The results suggest an increased sensitivity to at least one subtype of muscarinic receptor in Pb-treated rats.

Glutamatergic functioning also has been assessed by use of the drug discrimination paradigm. Rats chronically exposed beginning at weaning and tested as adults exhibited diminished responsiveness to an NMDA subtype receptor antagonist (Cory-Slechta, 1995b) but enhanced responsiveness to lower doses of NMDA (Cory-Slechta et al., 1996b). When exposure was limited to the period between birth and weaning, the diminished sensitivity to the NMDA receptor antagonist was less evident, but still present (Cory-Slechta, 1997b).

Thus, the drug discrimination paradigm appears to provide useful insights into the status of some neurotransmitter systems in chronically Pb-exposed animals. The reports cited above indicate an upregulation of dopaminergic, cholinergic, and glutamatergic receptors that are generally consistent with findings of diminished presynaptic function described earlier in this section of the current Lead AQCD. Nonetheless, this paradigm has some limitations. As all drugs in the cited studies were administered systemically, the results provide no evidence on brain regional sites of action. In addition, the chronic intermittent administration of the training drug has the potential to induce compensatory neuronal changes by itself, and thusly may mask or otherwise alter the manifestation of the effects of Pb exposure. Future use of this paradigm in Pb neurotoxicity studies must acknowledge this latter consideration.

Other Effective Behavioral Test Paradigms

Another test paradigm effectively utilized at least transiently to distinguish changes in chronically Pb-exposed animals is the repeated acquisition and performance schedule (Cohn et al., 1993). The purpose of this paradigm was to determine the selectivity of Pb-induced changes in learning, as distinct from nonspecific or performance effects, and to explore the

nature of the underlying error patterns contributing to any learning deficits. This schedule required completion of a sequence of three responses for reinforcement, with the correct sequence for the learning (i.e., repeated acquisition) component changing with each successive experimental session, while the performance component sequence remained constant across sessions.

The use of this schedule in animals chronically exposed to Pb beginning at weaning uncovered significant decrements in accuracy on the learning component, but not on the performance component, in Pb groups compared to controls (Cohn et al., 1993). A detailed analysis of subjects' behavior indicated that Pb exposure impaired learning by increasing perseverative responding on a single lever, even though such repetitive responding was not directly reinforced. In a subsequent study, dose-effect curves for the NMDA receptor antagonist MK-801 were determined in controls and animals tested in this paradigm in which chronic exposure began at weaning (Cohn and Cory-Slechta, 1993). The decline in learning accuracy and the increases in perseverative responding produced by MK-801 were attenuated by Pb exposure, and dose-effect curves relating MK-801 dose to changes in rates of responding were shifted to the right in exposed rats compared to control animals. These observations, therefore, demonstrated a subsensitivity of Pb-exposed animals to both the accuracy-impairing and response rate-altering properties of the antagonist. An additional investigation utilized the same Pb exposure protocol and administration of doses of NMDA as a receptor agonist to rats undergoing this test paradigm (Cohn and Cory-Slechta, 1994b). In control animals, NMDA was found to decrease accuracy of response in both the repeated acquisition and performance components of this multiple schedule and to suppress response rates as well. Lead exposure potentiated the accuracy-impairing effects of NMDA by further increasing the frequencies of errors and likewise potentiated the drug's rate-suppressing effects. Thus, as stated earlier in this section, the Pb-induced potentiation of the agonist effects and reduced sensitivity to the antagonist effects in this test paradigm are consistent with an increased density or some other upregulation of NMDA receptors in exposed brain tissue. In other work, Cohn and Cory-Slechta (1994a) were unable to distinguish any evidence of dopaminergic modulation of responding in this behavioral paradigm. Thus, the repeated acquisition and performance schedule proved valuable not only in providing a finer dissection of the animal's behavior, but in elucidating

important aspects of Pb neurotoxicity without some of the limitations inherent with drug discrimination or other behavioral test methods.

Summary

There is general agreement that the important factors in determining behavioral responses of Pb-exposed animals are (a) the nature of the test paradigm and its sensitivity to exposure and (b) the timing and intensity of the Pb exposure. Detailed analyses of responding have shown that Pb-exposed animals are less sensitive to the changing reinforcement contingencies that are integral to series of reversal tasks. They exhibit shortened initial periods of responding to the previously correct cue in combination with prolonged postperseverative learning phases for the new task. These have been proposed to be associative deficits based on deficiencies in learning new response contingencies. In addition, the impaired responding of Pb-exposed animals in vigilance tasks has been attributed to deficiencies in sustained attention and an increased reactivity to errors.

Other test paradigms such as drug discrimination and repeated acquisition/performance tasks have provided useful assessments of the integrity of CNS neurotransmitter systems in Pb-exposed animals. Evidence from both paradigms has been in general agreement in indicating up-regulated neurotransmitter receptor systems. The timing of Pb exposure is critically important in determining the response to cocaine, and the potentiating action of perinatal Pb exposure is of potential importance for public health purposes.

5.3.1.6 Lead-Induced Changes in Cellular Development and Disposition of the Metal

Alterations in cellular differentiation and morphology can be important structural neuronal and glial components of the manifestations of Pb neurotoxicity. While these issues have not been thoroughly addressed by research investigations, there have, nonetheless, been important observations made. This subsection reviews studies concerned with various aspects of this topic.

Lead Exposure and Neural/Glial Progenitor Cells

Studies of the effects of Pb exposure on neural and glial progenitor cells are recent occurrences in the field of Pb neurotoxicity research. Chronic exposure in rats begun at postnatal

day 25 was found to significantly decrease proliferation of new cells in the dentate gyrus compared to the extent of this process in control animals (Schneider et al., 2005). Other workers initiated Pb exposure at birth and determined that continuous exposure to adulthood reduced the total number of labeled cells in the hippocampal dentate gyrus at 28 days, but not 24 h, after the last administration of a DNA synthesis marker (Gilbert et al., 2005). Rats whose exposure was terminated at weaning exhibited no changes in cellular labeling or survival, indicating that chronic exposure reduces the capacity for hippocampal neurogenesis.

Studies have also been conducted to investigate the effects of exposure on glial progenitor cells. Deng et al. (2001) examined cultured oligodendrocytes and their progenitor cells acutely exposed to Pb^{2+} in vitro; they observed an exposure-induced delay in the differentiation of the progenitors, and that the progenitor cultures were more sensitive to Pb^{2+} than the mature oligodendrocytes. These findings suggested interference with the timely developmental maturation of the progenitor cells. A subsequent study found that a low concentration of Pb^{2+} in vitro inhibited proliferation and differentiation of these progenitors without affecting cell viability (Deng and Poretz, 2002). Proliferative capability was decreased and cell-intrinsic lineage progression was inhibited at a late progenitor stage. Thus, acute Pb^{2+} suppresses both the proliferation and differentiation of these cells.

Lead Exposure and Neurite Outgrowth

Neurite initiation is known to be highly sensitive to neurotoxic compounds and has been the focus of studies examining morphological alterations caused by exposure to Pb^{2+} in vitro. Kern and Audesirk (1995) found that 100 nM Pb^{2+} inhibited neurite initiation in cultured rat hippocampal neurons and, on the basis of results with kinase inhibitors, concluded that this occurred by inappropriate stimulation of protein phosphorylation by Ca^{2+}-calmodulin-dependent or cyclic AMP-dependent protein kinases, possibly through stimulation of calmodulin. Intracellular free Ca^{2+} concentrations were not altered by up to 48 h exposure to nominal 100 nM Pb^{2+}, leading these workers to propose that the stimulation of the above kinases or calmodulin were not via increased Ca^{2+} but, instead, were attributable to intracellular Pb^{2+} concentrations. Evidence of Pb^{2+}-induced inhibition of neurite outgrowth is in general agreement with observations made after chronic exposure to Pb employing in vivo models. Cline et al. (1996) employed an exposure protocol of 0.1 nM–100 μM nominal Pb^{2+} for 6 weeks localized to the

retinotectal system of frog tadpoles, and observed a severely reduced area and branchtip number of retinal ganglion cell axon arborizations within the optic tectum at nM Pb^{2+} concentrations. Reuhl (1989) exposed primates to 2 mg lead/kg/day from infancy to 6 years of age and found that neuronal volume density was reduced in primary visual area V1 and in visual projection area V2 compared to a group exposed to 25 μg lead/kg/day. Moreover, a relative decrease in the number of arborizations among pyramidal neurons in both areas V1 and V2 was observed in the higher dose group. Thus, there is good correspondence between reports that acute Pb^{2+} exposure in vitro and extended exposure in animal models in vivo results in diminished neuronal growth and differentiation at Pb levels of apparent environmental relevance. Studies employing intact animals have not progressed to investigation of specific cellular mechanisms underlying these effects.

Lead Exposure and Neural Stem Cells

Given considerable contemporary interest in the use of neural stem cells to treat various neurological diseases, the efforts of Huang and Schneider (2004) to examine the actions of exposure to Pb^{2+} in vitro on these cells is worthy of note. Lead exposure produced no effect on neurosphere viability, but, it did cause a significant dose-dependent inhibition of proliferation. In addition, the number of neurons differentiated from Pb^{2+}-exposed neurospheres was significantly decreased from control, as were the number of oligodendrocytes obtained. However, Pb exposure increased the number of astrocytes obtained. These observations suggest an important Pb^{2+}-induced influence on stem cell proliferation and differentiation that has public health relevance to prenatal metal exposure.

Accumulation of Lead in Brain

Most studies of neurotoxicity involving chronic Pb exposure now report blood and brain Pb concentrations to quantify exposure magnitude and/or as quality control measures. Thus, a sizable amount of data is available on general aspects of Pb toxicokinetics. While brain Pb values vary monotonically with blood Pb concentrations and exposure levels, steady state accumulation/washout times are longer in tissue and are dependent on exposure magnitude and duration. The half-time for the decline of Pb in brain tissue when exposure is terminated is on

the order of 10 days to 2 weeks, while the value for blood leads would be a matter of a few days (Lasley, unpublished observations).

The speciation and distribution of Pb in brain tissue is largely unknown except for indirect indications that only a small fraction of the divalent cation is present in tissue in extracellular fluid in the free ion state. The existence of a lead-binding protein in brain cytosol was reported by Goering et al. (1986) and was invoked to explain the relatively weak inhibition by Pb^{2+} of brain δ-aminolevulinic acid dehydratase activity. But the binding protein was not fully characterized or identified. Pb^{2+} is known to bind to various intracellular Ca^{2+} binding proteins, such as calmodulin, PKC, and synaptotagmin, particularly those with a C2 domain (Sun et al., 1999), but these are low-volume sources that have been studied for their functional importance and would not serve any kind of tissue metal storage function.

Pb^{2+} appears to be taken up into cultured cells by multiple ion channel-based mechanisms including influx through channels activated by depletion of intracellular Ca^{2+} stores, non-L-type Ca^{2+} channels, and NMDA receptor-associated channels (Kerper and Hinkle, 1997; Mazzolini et al., 2001). Astroglia are well known to act as a Pb sink and, in culture, accumulate up to 24 times more of the metal than neuronal cells (Lindahl et al., 1999); and there is evidence that glutathione may regulate Pb uptake into these cells. Only recently has one astroglial protein been identified—a molecular chaperone in endoplasmic reticulum (Qian et al., 2000, 2005), glucose-regulated protein (GRP78). Intracellular levels of this protein are increased in cultured astroglia during one week's exposure to Pb^{2+}, suggesting that this protein is a component of the intracellular tolerance mechanism that handles high intracellular Pb accumulation through a direct interaction. GRP78 depletion significantly increased the sensitivity of cultured glioma cells to Pb^{2+} as indicated by the generation of reactive oxygen species. Thus, it appears that Pb^{2+} directly targets the protein and induces its compartmentalized redistribution, enabling it to play a protective role in Pb neurotoxicity.

5.3.1.7 Integration of Research Findings

It is evident that the Pb neurotoxicity literature is broad and varied and that many valuable observations have been made over the last 20 years. Nevertheless, a few general conclusions are in order so as to help integrate and concisely summarize evidence in at least a few focused areas.

- Lead-induced impairments in glutamatergic neurotransmission appear to underlie the deficits in synaptic plasticity and in learning or acquisition behavior. Cholinergic neurotransmission also serves an important role in impaired learning in exposed subjects, while deficits in dopaminergic function are manifested as alterations in rates of responding or incentive motivation. Exposure-related alterations in structural plasticity appear to be based on interference with Ca^{2+} signaling and/or glutamatergic transmission.
- There is little if any support in the Pb neurotoxicity research community for the notion of thresholds for any of the toxic mechanisms that have been addressed in this section of the document. With the pressure to reduce experimental group sizes to the minimal number necessary and the unspecified notion that rats are somewhat more resistant to Pb than children, most studies performed with in vivo models report blood Pb values in the range of 15 to 35-40 µg/dL. Moreover, in view of the complex and undefined speciation equilibria and distribution of Pb in physiological milieus, there is no way to directly relate a blood Pb value to the levels of free Pb^{2+} ion or to any other complexed active form of the metal, either in extracellular or intracellular fluids. Generally accepted estimates of the free Pb^{2+} ion concentrations produced in brain extracellular fluid by environmentally relevant exposures fall in the low nanomolar range.
- Susceptibility factors for Pb neurotoxicity are poorly defined in laboratory animals, and, thus, have not been studied. A compelling rationale for their investigation has not been provided.

5.3.2 Neurotoxicological/Neurobehavioral Effects of Lead in Humans

This section is divided into three sub-sections, based upon age and exposure scenarios. The sub-sections include (1) children with blood lead levels above and below 10 µg/dL, (2) adult manifestations of neurotoxicity and other disease states as a result of excessive exposure to lead as children, and (3) adults who were exposed to "ambient" levels of lead. In each of these sub-sections, wherever possible, discussion is focused on biochemical markers, bioclinical markers, and reversibility of lead's neurotoxic effects. In addition, for each of the groups cited above, vulnerability to the neurotoxic effects of lead is considered. Topics in this area include developmental toxicology and growth and development in children. For children and adults, other aspects of vulnerability are considered, such as socioeconomic status, nutrition, and genetic polymorphisms. Based upon the body of studies discussed in each sub-section, it is reasonable to draw conclusions relating to dose-response paradigms and clinical extensions of epidemiological data to individual children.

5.3.2.1 Effects of Lead in Young Children to Mid-Adolescence

Since EPA's publication of the Air Quality Criteria Document and Addenda in 1986-1990{EPA-600/8-83/028aF(1986); EPA/600/8-89/049F(1990)}, major studies with new and critical information have substantially extended previous hypotheses expressed by the EPA. These new data are the primary and major departure from EPA's earlier lead criteria reviews. It is now recognized that lead has adverse effects on the developing central nervous system of young children and that these effects on cognition and behavior persist (at least) into the school-aged years and beyond into mid-adolescence. While causal conclusions about effects of lead exposure on cognitive development are made with caution, collectively, the nature and abundance of the evidence is clear and compelling. This new information causally links detrimental effects of lead on behavior and cognition at blood lead levels both above and below 10 µg/dL. Compared to the earlier EPA documents (cited above), there is solid evidence for detrimental effects of lead on neuropsychological functions such as fine motor skills, visual-spatial, and executive functioning and attention in large groups of children. Apparently, there is no threshold below which lead is without adverse effects on the central nervous system of young children to mid-adolescence. This conclusion is also related to dose-response paradigms of lead in children, as well as to extending the results of epidemiological studies to individual children.

Biomarkers

There are three generally recognized types of biomarkers (National Research Council [NRC], 1993; Lanphear and Bearer, 2005). The first of these is relevant to quantifying exposure and, thus, the internal dose of lead. This subject is covered in Section 6.1. The second type of biomarker focuses on *effects*. Biomarkers of effect are biochemical, physiological and/or clinically measurable alterations in normal functioning that reflect an impairment of health or a specific disease. Effects of this nature include early subclinical effects of lead that are of value in quantifying human health risks; as a result, such effects can lead to an understanding of mechanisms of lead toxicity at the cellular and organ level. An example of the latter are the impacts of lead on heme synthesis, which have been extensively studied as discussed in Section 5.2. Other examples include (in children) effects on the electrophysiology and architecture of the brain. In contrast, biomarkers of *susceptibility* or *vulnerability* are native,

inherent or acquired situational characteristics that alter the responses of children (and adults) to lead exposure. Examples of these biomarkers include socioeconomic status, nutrition, developmental aspects of brain functions, and genetic polymorphisms. Ultimately, these biomarkers, collectively, as a constellation, are considered to be contributing factors that assist in determining whether the cognitive effects of lead exposure in children are reversible.

Biochemical Biomarkers

In the AQCD of 1986, reported studies in lead-poisoned children (blood lead levels of 12-120 μg/dL) revealed an inverse correlation (−0.88) between the entire range of blood lead concentrations and plasma levels of the vitamin D hormone (1, 25-dihydroxyvitamin D) in 177 children from 1-16 years of age (Rosen et al., 1980; Mahaffey et al., 1982). These results suggested that lead impairs the biosynthesis of the Vitamin D hormone; and, as a result, calcium absorption, and possibly that of lead, could be inhibited. Because of 1,25-dihydroxyvitamin D's roles in multiple cellular functions, including the calcium-messenger cascade, development and proliferation of multiple cell types, these clinical observations have substantial implications (NRC, 1993). These data from clinical studies were supported in experimental animals (Smith et al., 1981). In animals fed a low calcium or phosphate diet, oral administration of 0.82% lead, as the acetate, yielded plasma levels of 1,25-dihydroxyvitamin D that were substantially reduced. This effect of lead on circulating 1,25-dihydroxyvitamin D disappeared when either a high calcium or phosphate diet, including lead, was administered. Moreover, intestinal lead appeared to block the absorption of calcium in response to administration of 25-hydroxyvitamin D and 1,25-dihydroxyvitamin D, although there was no influence on calcium mobilization neither from bone nor of mineralization of rachitic bone (Smith et al., 1981).

More recently, in animals and children, this relationship has been examined further. Chicks fed lead concurrently on a low calcium diet replicated the findings in animals and children noted above. However, chicks fed a calcium sufficient diet coupled to dietary lead, failed to exhibit decreased plasma levels of the vitamin D hormone (Fullmer and Rosen, 1990; Fullmer, 1995, 1997). Similar findings were reported in lead poisoned children with adequate dietary intakes of calcium as those in experimental studies (Koo et al., 1991). Compared to the 1980s, when dietary intakes of calcium were marginal (at best) in inner-city children, recent estimates of dietary calcium intakes in inner city children meet or exceed recommended daily

requirements of 1000 mg/day or greater (Markowitz et al., 2004). Thus, in calcium-sufficient children, plasma concentrations of 1,25-dihydroxyvitamin D are not biomarkers of lead's effects on the vitamin D endocrine system. Nonetheless, in at-risk populations of children, whose dietary intakes of calcium are suboptimal, apparent biosynthesis of the vitamin D hormone, evidenced by decreased circulating levels of the hormonal form of the vitamin, 1,25-dihydroxyvitamin D, is expected.

Tang and co-workers (1999) assessed 244 infants on the Brunet-Lezine Scales at 9 months of age to evaluate possible relationships between cord blood lead levels and plasma concentrations of 5-hydroxyindoleacetic acid (5-HIAA) and homovanillic acid (HVA) at 9 months of age. Cord blood lead concentrations were in the range of 2.5-7 μg/dL. At 9 months of age, 5-HIAA and HVA were negatively correlated with blood lead values and with all the neurodevelopmental functions, except for language, on the Brunet-Lezine Scales. The negative correlations between the serotonergic system, coupled to blood lead levels, were found in global scores, sociability, and coordination on the Brunet-Lezine Scales. Although further confirmation of these results is needed, these findings are consistent with the findings in experimental studies linking lead effects to impairments in neurotransmission (Section 5.3).

Clinically Oriented Biomarkers of Effect

Very little information was available in the time frame of 1986-1990 relating to clinically oriented biomarkers of lead's effects on the central nervous system of children. Currently, there is a substantial body of knowledge which focuses on the functional status of the brain in excessively lead-exposed children. This new information includes functional (Bhattacharya et al., 1993; Rothenberg et al., 1995) and electrophysiological (Otto and Fox, 1993; Burchfiel et al., 1992; Poblano et al., 2001; Rothenberg et al., 2000) studies. The most relevant studies of functional and electrophysiologic data, relating to spectral analyses of EEGs, brainstem auditory-evoked potentials or responses (BAEP and BAER) are those based upon prospective designs of a well characterized cohort of children. Another line of important new information is based upon results from assessments of the biochemical and anatomical functions of the central nervous system in lead-exposed children carried out by magnetic resonance spectroscopy (MRS)(Trope et al., 2001; Meng et al, 2005). Some of these studies link observed results to cognitive impairments summarized in the general introduction to this chapter. Although there may be

some overlap in these reported studies, for the purposes of this discussion, studies are categorized as functional, electrophysiologic and biochemical-anatomical.

A functional assessment of postural equilibrium was carried out in 109 children (in the Cincinnati Prospective Study) at 5.8 years of age when the mean blood lead was 5.8 µg/dL (the geometric mean blood lead for the first 5 years of life was 11.9 µg/dL). Postural sway was quantitated with a microprocessor-based platform. A negative correlation was found between blood lead levels and vestibular/proprioceptive systems, suggesting that lead exposure has detrimental effects on posture and balance. Although these data may have potential relevance to psychomotor deficits observed in lead poisoned children, this methodology has not been further developed, and its potential implications to psychomotor skills in children have not evolved (Bhattacharya et al., 1993).

In the Mexico City Prospective Study, analyses of acoustical cries was carried out in a subset of healthy babies at 2, 15, and 30 days of life. The mean maternal blood lead at 12 weeks of pregnancy was 8.2 µg/dL, and the mean cord blood lead level was 7.8 µg/dL (range: 1-38 µg/dL). The percent nasalization, produced by raising the velum of the velopharynx, decreased progressively over the cord blood lead range of 4-40 µg/dL, and the number of cries were inversely related to cord blood lead levels over the same range. In a subset of the babies, decreased nasalization was related to increased BAEP-evoked latencies. These findings suggested that altered baby cries and auditory functions may be associated with developmental delays affecting early communication. However, validation of infant cries as a predictor of subsequent infant development has not yet evolved.

Burchfiel et al., (1992) studied a subset of the Philadelphia children reported on by Needleman et al. (1979). The method used was brain electrical activity mapping (BEAM) coupled to spectral analysis for each individual electrode. Nineteen children in the uppermost 10th percentile for dentine lead (>24 ppm) were compared to children in the lowest 10th percentile (<6 ppm). The spontaneous resting EEG of the high lead children (n = 19) had higher percentages of low frequency delta activity and reduced percentages of alpha activity compared to the lower lead group of 12 children. Qualitatively, these EEG changes are similar to those observed in acute-severe lead poisoning, and, generally, such findings of diffusely increased slow frequency activity and reduced alpha are commonly found in toxic encephalopathies. These results, which are qualitatively similar to results from the Mexico City Prospective Study,

in which BAER were employed in a different age group, indicate that lead may induce neuropathological effects in a dose-response manner over a continuum of exposure (see Section 5.3).

In the Mexico City Prospective Lead Study, 100-113 5 to7 year olds underwent testing by BAER (Rothenberg et al., 2000). The mean blood lead level at 5 years of age was 8 μg/dL. The results indicated that intervals I-V and III-V intervals of BEAR recorded at 5-7 years of age were related to maternal blood lead at 20 weeks of pregnancy, when the geometric mean blood lead was 7.7 μg/dL (range: 1-30.5 μg/dL). This specificity of the lead effect suggests that the CNS is exquisitely sensitive to lead when auditory structures are undergoing rapid development, and this effect appears to persist to 5 or more years later. As indicated in this study, lead-related alterations in auditory brain stem function may underlie verbal deficits in lead-exposed children, as well as impair auditory functions observed in lead-exposed animals (Section 5.3.1). Also, it becomes increasingly important to examine functional deficits in hearing and language development that may be associated with postnatal lead exposure.

Nine to ten year-olds in the Mexico City Prospective Study were evaluated by determining relative theta activity across the scalp. These results, together with life time blood lead concentrations, were assessed by multiple regression models (Poblano et al., 2001). The most significant increases in theta power were associated with blood lead levels (geometric mean: 10.3 μg/dL) measured between 54 and 72 months of age. Spatially weighted regression showed that there was a significant anterior-posterior gradient in the lead-induced increase in relative theta activity associated with postnatal blood leads at 54-72 and 78-96 months. These lead effects occurred at an age during which relative theta power reaches its developmental maximum and then starts to decrease. These data have critical implications in understanding the neurotoxic and developmental impacts of lead exposure. If theta waves continue throughout childhood as one of the most dominant CNS rhythms, this could qualify as a developmental disorder and an "EEG soft sign." Stated differently, persistence of theta activity reflects an 'immature" EEG pattern and/or brain injury. Neuropsychological testing concurrent with this electrophysiological methodology could add important information connecting direct measures of theta activity to CNS development indexed by neuropsychological outcomes (Lidsky, 2003, 2005).

Using MRS, two studies (Trope et al., 2001; Meng et al., 2005) have provided new and important information. Both studies employed MRS with N-acetylaspartate (NAA). MRS has the capability to monitor brain metabolism by detecting NAA, a metabolite that is known to decrease during processes involving neuronal loss. Thus, this methodology provides both biochemical and anatomical information directly related to the neurotoxic effects of lead on the CNS. Trope et al. (2001) studied 16 lead-poisoned children (mean blood lead: 39.9 μg/dL; range: 23-69 μg/dL) who had a mean age of 8 years, 9 months. All of these children received medical attention before 5 years of age. The latter group was compared to 5 children (blood lead levels <10 μg/dL) who had a mean age of 8 years, 6 months. Both groups of children had normal MRIs. The lead-exposed group of children had significant reductions in NAA/creatine and phosphocreatine ratios in frontal gray matter compared to the nonexposed group. Review of medical records in the lead-exposed group failed to reveal an alternative or contributing etiology that could explain this demonstration of brain damage by MRS. These findings, in the regions of the frontal lobes, which are responsible for attention, executive functions and impulse control, are likely to be relevant to neurotoxic outcomes in lead exposed children, who may exhibit impairments in these areas.

Using very similar techniques, Meng et al. (2005) evaluated 6 lead-exposed children (mean blood lead: 37 μg/dL) who lived near a lead-recycling industry for a period of at least 5 years. These children had never been chelated. The lead-exposed children were compared to 6 non-lead exposed children who had blood lead levels <10 μg/dL. On the Wechsler scale, the control children had a Full Scale IQ of 101 compared to 81 in the excessively exposed group. MRIs were normal in both groups of children. These data parallel those reported by Trope et al., (2001). However, Meng et al. (2005) reported decreased levels of NAA in the lead-exposed group in four brain regions: the left and right frontal areas and the left and right hippocampus. This study also found that MRS metabolites in the lead-exposed subjects were significantly reduced as compared with controls, thereby suggesting interference with neuronal functioning after lead exposure.

Collectively, these electrophysiological and biochemical-anatomical data can assist in providing an understanding of the neurotoxicity of lead and neurophysiological outcomes in relatively “low” and “higher” level childhood lead exposure. Collectively, these data also appear

to provide evidence that lead interferes with the hard-wiring and differentiation of the central nervous system in children.

Vulnerability and Susceptibility

The unique susceptibility of children to the adverse health effects of lead were recognized previously by EPA in 1986-1990. Some of these aspects included the specific behaviors of children, including their metabolism of lead, physiological considerations that separate children from adults, greater potential absorption of lead per square meter of body surface, hand-to-mouth activity, and prevalence of nutritional factors that can enhance the absorption of lead from the GI tract.

Since 1986-1990, an enlarged database is now available to construct a somewhat wider approach to understanding not only new information relating to children's susceptibility, but also furthering characterizing interindividual variability as related to manifestations of lead's adverse health effects in children. This section includes topics of developmental toxicology, growth and development, economic status, nutritional aspects of lead and children, and, finally, genetic considerations of children with possibly a biologically based genetic character interacting with exterior environmental realities. Considerations in this section also delve into risk assessment focused on some child-specific factors that affect health outcomes in populations as well as individual children.

Moreover, as a general principle of toxicology and neurotoxicology, it is recognized that a variety of factors can either enhance or decrease an individual's sensitivity to toxic exposures of lead. Besides individual children, there are factors that modify the selective neurotoxic responses of subgroups of children. Some of these variables that increase a child's vulnerability are discussed below.

Developmental Toxicology

In addition to child-specific factors detailed above, it was concluded previously in 1986–1990 that the critical window of adverse health effects of lead in children was at less than 3 years of age, as briefly mentioned below and in greater detail in Section 6.3. This suggestion of age should be extended to children in their school-aged years to mid-adolescence (Chen et al., 2005; Ris et al., 2004; Dietrich et al., 2004) and into the adult years as well (Rice and Barone, 2000).

Developmental toxicity is usually defined by the occurrence of adverse effects on the developing organism prior to conception and/or during prenatal and postnatal development. Manifestations of developmental toxicity include death to the developing organism, structural abnormalities, impaired growth, and functional deficiencies. Moreover, developmental exposures can result in adverse health effects prenatally, postnatally, in childhood, into school-aged years, and into adult age groups to the elderly (Selevan et al, 2000; Weiss, 2000; Rice and Barone, 2000). An important concept in risk management is to identify, whenever possible, developmental windows for evaluating dose-response relationships. Moreover, in risk management, identification of critical windows is aimed at recognizing especially susceptible sub-groups within the general population to provide specific interventions (Selevan et al., 2000). Information on critical windows of development is needed to assess real and potential environmental health risks (Weiss, 2000).

To protect children's health, it is necessary to understand their unique sensitivity to environmental toxicants, and, to further this understanding, functions of risk and exposure must be considered (Faustman et al, 2000). Risk is defined as the probability of an adverse outcome as a function of exposure and toxicity. It is evident that in development of the CNS, unidirectional inhibition at one developmental stage can cause substantial alterations in subsequent processes. In addition, stages of development occur in temporally distinct time frames across regions of the brain (Rice and Barone, 2000; Weiss, 2000). As a result, the CNS has a very limited capability to compensate for cell loss or other injury (Rice and Barone, 2000). Thus, exposure criteria should be based on information relevant to predicting risks and to accounting for toxicokinetic differences that occur during different stages of development (Faustman et al., 2000).

Characterization of critical time frames of development in children are based, in large part, on the results of experimental studies discussed in Section 5.3.1. Initially, the critical time frame for adverse effects of lead on CNS development was considered to be in children <3 years of age (Bellinger et al., 1991, 1992) in that blood lead levels at 2 years of age were correlated with cognitive impairments at 57 months and 10 years of age. However, as indicated in Section 6.3, the age range for time windows for lead's adverse effects on the CNS has been significantly extended to school-aged children, into adolescence, and into adulthood (Dietrich et al., 1993; Tong et al., 1996; Wasserman et al., 2000; Canfield et al., 2003, 2004; Chen et al., 2005;

Lanphear et al., 2000, 2005). Moreover, recapitulation of synaptogenesis in the form of synaptic plasticity is modified by experience and the environment as children become adults and age into the elderly life phase (Rice and Barone, 2000). This concept provides a toxicological framework for identifying latent or persistent expressions of childhood lead exposure in adults as "growing into a lesion" (Ris et al., 2004) or magnification of an earlier insult with aging (Rice and Barone, 2000). This toxicological recognition of latent or persistent expressions of childhood exposure in adults forms the basis of Section 5.3.2.2, discussed below. Additional areas of concern for children related to risk assessment include consideration of lead's deleterious effects on somatic growth, socioeconomic status, nutritional correlates of lead exposure and interactions between biologically inherent genetics and the external environment (discussed below).

Growth and Development

In the Supplement to the 1986 Addendum (EPA/600/8-89/049F), early results from prospective studies in Cincinnati, Boston, Port Pirie, and Yugoslavia were noted in terms of lead's effects on perinatal and postnatal growth and development. However, evidence regarding physical growth effects related to prenatal or early postnatal exposure were inconsistent. Limitations in these early data from prospective studies included definitions of the length of gestation, racial makeup, maternal age, sample sizes, and levels of lead exposure. It appeared likely that prenatal lead exposure did pose a potential hazard to the developing fetus as related to reduced gestational length and possibly other aspects of fetal growth. It proved difficult, however, to define a definite dose-response relationship for fetal outcomes, although there were some indications that pointed to adverse effects on the fetus at blood lead levels of 10-15 µg/dL.

More recently reported data have extended assessments of impacts of lead on early postnatal outcomes (birth weight, early weight gain to 1-month of age, birth length and head circumference) to measurements of maternal bone lead post-delivery by K-XRF. Additional studies have analyzed national data (NHANES III) in terms of the growth patterns of children 1 to 7 years of age. From the same national survey, other studies have focused on sexual maturation as a function of blood lead concentrations.

Gonzalez-Cossio et al. (1997) assessed the possible relationship of blood lead levels in cord blood and maternal bone lead to birth weight in Mexico City. Two hundred seventy-two mother-infant pairs were studied, and the cord and maternal blood lead levels were, on average,

7.1 and 8.9 µg/dL, respectively. Tibial lead (not patellar lead), measured 1-month postpartum by K-XRF, was the only marker related to birth weight, such that, at the highest quartile of bone lead (15.15 µg/g bone mineral), infants were, on average, 156 grams lighter at birth. Although these data appear to extend previously reported information from prospective studies, some caution in the interpretation of these data is indicated: 10% of tibial lead values and 13% of patellar lead levels were below the instrument's detection limit (as defined by the authors), but these negative values were included in the statistical analysis. By necessity, the study design was cross-sectional, because, for ethical reasons, bone lead measurements are precluded during pregnancy due to radiation exposure, and, for unexplained reasons, there were no statistically significant relationships with blood lead values in mothers or infants.

Sanin et al. (2001) studied a similar population in Mexico City comprising 329 mother-infant pairs. The umbilical cord blood lead (mean) was 6.8 µg/dL and the mean value for the infants was 5.6 µg/dL at 1-month of age. A 10 µg/dL increase in infant blood lead levels at 1-month was associated with a 142 gram decrease in weight gain. Thus, lead exposure postnatally had adverse effects on early perinatal weight gain. In addition, maternal patellar lead at 1-month was negatively associated with weight gain as well. The important finding in this study was the inverse correlation between postnatal blood lead and weight gain. However, the significance of maternal patellar lead by K-XRF is limited by the large standard deviation in the measurement of patellar lead (15.2 µg/g bone mineral) and a revised statistical method for calculating bone leads, which was not delineated in this article. Moreover, the failure of maternal blood lead concentration to predict early or subsequent birth weight was unexplained.

In the third of this series of reports from Mexico City, 233 mother-infant pairs were evaluated as described above, but the outcome measures were birth length and head circumference. These results showed that bone lead biomarkers (tibia and patella) were associated positively and significantly with maternal and umbilical cord lead. However, only patellar lead was significantly and negatively associated with birth length and head circumference. These associations were independent of maternal venous blood lead levels, umbilical cord lead levels, and other predictors of birth size, including birth weight. Similar concerns apply here, as those above, relating to K-XRF measurements. Collectively, except for the relationship between postnatal blood lead concentrations and decreases in weight gain (Sanin

et al., 2001), the cumulative findings relating to perinatal and postnatal outcomes, as indexed by bone lead values, fail to provide a consistent or readily interpretable set of conclusions.

Evaluation of 4391 children 1-7 years old was carried out using the nationally representative data from NHANES III. This population study included non-Hispanic white, non-Hispanic African-American, and Mexican-American children, and the outcomes measured were stature, head circumference, weight and body mass index (Ballew et al., 1999). Blood lead levels were significantly and negatively related to stature and head circumference, yielding a predicted decrease of 1.57 cm in stature and a 0.52 cm decrease in head circumference for each 10 µg/dL increase in blood lead values. There was no statistically relevant association between blood lead and weight and body mass index. These robust findings are of considerable importance, because the observations are very similar to those reported earlier for NHANES II by Schwartz et al. (1986). Thus, although blood leads declined substantially in the United States over two decades (NHANES II vs.NHANES III), lead exposure at considerably lower levels continued to affect the growth of some children. Stated differently, there was no attenuation of the negative association between blood lead levels and indices of growth in children despite a substantial decrease in national blood lead values in young children. Collectively, these cross-sectional national surveys (Schwartz et al., 1986; Ballew et al., 1999) indicate the following negative associations with blood lead values: a 1.0-1.5 cm decrease in stature and a 0.50 cm decrease in head circumference coupled to modest increases in blood lead levels within the range of blood lead levels in children in the United States. Over the past two decades, these data provide the most compelling data sets related to the adverse effects of lead on growth patterns in young American children.

Two studies, utilizing NHANES III data, have measured pubertal development, as related to blood lead concentrations, to determine whether sexual maturation may be affected by current environmental lead exposure (Wu et al., 2003; Selevan et al., 2003). In the study by Wu et al. (2003), pubic hair and breast development were evaluated in 1,706 8–16 year-old girls, and information on menarche was delineated in 1,235 girls 10–16 years of age. The blood lead range was 0.7–21.7 µg/dL. This population was categorized into three groups, according to blood lead values: 0.7–2.0, 2.1-4.9, and 5.0–21.7 µg/dL. Sexual maturation markers were self-reported attainment of menarche and physician-determined Tanner stage 2 pubic hair and breast development. The results indicated that girls who had not yet reached menarche or stage 2 pubic

hair had higher blood leads than those girls who had. Negative relationships were found for blood leads with attainment of menarche and stage 2 pubic hair after adjusting for covariates; no relationships were evident for breast development.

Selevan et al. (2003) studied a subset of girls (8-18 years old) from NHANES III that included 600 non-Hispanic white, 805 non-Hispanic African-American, and 781 Mexican-American girls, who, collectively, had a geometric mean blood lead level of 3 μg/dL. For all girls who had blood lead levels of 3 μg/dL compared to those whose blood leads were 1 μg/dL, the higher lead group had a significant decrease in height after adjustment for confounders. Also, in the higher lead group of girls, there were significant delays in breast and pubic hair development, especially in non-Hispanic African-American and Mexican-American girls. Of the latter two groups, the most profound delays were observed in non-Hispanic African-American girls. The delays in reaching Tanner stages 2, 3, 4, and 5 (in all girls) was associated with those whose geometric mean blood lead was 3 μg/dL (as compared to all girls whose geometric mean blood lead level was 1 μg/dL) were 3.8, 5.3, 5.8, and 2.1 months, respectively, for breast and pubic hair development. There were no significant delays found in non-Hispanic white girls. These findings, within a narrow blood lead range, indicate that environmental lead exposure can delay growth and pubertal development in girls. Thus, analyses of these national surveys in children 1-7 years old and in girls 8-18 years old provide strong evidence for adverse effects of lead on the growth of young children and adolescent girls at blood lead concentrations commonly found in the U.S. population today.

Socioeconomic Status

In the U.S. EPA's Supplement to the 1986 addendum (EPA/600/8-89/049F), very little information was discussed relating to socioeconomic status (SES) and the vulnerability of children to lead exposure and resulting deficits in cognitive skills. Primarily as a result of analyses of NHANES III, the importance of SES has reached its appropriate focus and attention. Additional peer-reviewed articles have also contributed to now well-documented interactions between SES and children's vulnerabilities to the neurotoxic effects of lead.

A child's SES clearly has an important influence on the possibility of lead exposure in young children. Disadvantaged children may have an already compromised neuropsychological status that is further impaired by the toxic effects of lead. Although the exact mechanisms of the

impact of SES on lead's neurotoxic effects on the central nervous system are unknown, poverty, pre-1960 housing in segregated communities, ethnicity, and nutritional deficiencies, collectively, can contribute substantially to increased vulnerability of individual children and subgroups of children. The peer-reviewed literature, discussed in this section and in the following section, provides support for these conditions contributing to children's susceptibility to the toxic effects of lead.

An analysis of the early phase (1988-1991) of NHANES III was carried out by Brody et al. (1994), in which 13,201 persons from 1 year of age through elderly adults were assessed via a multiple stage probability design. It was found that the prevalence of elevated blood lead levels for children in low-income families (16.3%) was about four times higher than the prevalence for children with high family incomes. Non-Hispanic African-American children from low-income families had the highest proportion of elevated blood lead values (28.4%).

A comparison of results from NHANES II and NHANES III (Pirkle et al., 1994) extended data reported earlier (Brody et al., 1994). From 1976 to 1991 (NHANES II vs. early NHANES III) there was an overall decline in all children 1-5 years old from 15.0 to 3.2 μg/dL (geometric means). For non-Hispanic white children, the decline was from 13.7 to 3.2 μg/dL, whereas for non-Hispanic African-American children, the decline was from 20.2 to 5.6 μg/dL. Income levels were based upon those previously determined by the U. S. Census Bureau: income level was defined by the poverty-income ratio (PIR), so that the total family income was divided by the current poverty threshold. PIRs were divided into three categories: low ($0 < PIR < 1.30$; mid-range ($1.30 < PIR\ 3.0$) and high ($PIR > 3.00$). Based upon PIRs, it is noteworthy that mean blood lead levels decreased by 60% (24.0 to 9.7) for African-American children from low-income families living in central cities with populations of 1 million or more. The latter value for low-income African-American children was about 3 times the mean value for non-Hispanic white children.

An analysis of phase 2 of NAHANES III showed that it becomes increasingly evident that SES factors, including sociodemographic factors, are closely related to average blood lead concentrations in young children (Pirkle et al., 1998). By phase 2, the average blood lead in all children 1–5 years old was 2.7 μg/dL. The prevalence of elevated blood lead levels in African-American children living in pre-1946 housing was 21.9%; the prevalence in all children of low-income families living in pre-1946 housing was 16.4% (demographic status was

determined by U.S. Department of Agriculture codes populations according to proximity to major metropolitan areas). Low incomes among all ethnic groups, defined by PIRs, were significantly associated with higher blood leads (details of these data were not provided in the article). It is reasonable to conclude that U.S. children, based on ethnicity, housing age, and income, are disproportionately exposed to excessive levels of lead in their environments.

SES was further considered in NHANES III (1988-1994) based upon age and blood lead concentrations (Bernard and McGheehin, 2003). Overall, 25.6% of children 1-5 years old had blood leads equal to or greater than 5 μg/dL; but most of these children (76%) had blood leads less than 10 μg/dL. Of those children who had blood leads greater than 5 μg/dL, 46.8% were non-Hispanic African-American compared to 18.7% non-Hispanic white children. Housing status also played a significant role: 42.5% of children who had blood leads greater than 5 μg/dL lived in pre-1946 housing, 38.9% lived in housing built between 1946-1973, and 14.1% of children in this blood lead group lived in housing built after 1973. Compared to non-Hispanic white children, African-American children were 3 times more likely to have blood leads greater than 5 μg/dL, 7 times more likely to have blood leads of 10-20 μg/dL, and 13.5 times more likely to have blood leads equal to or greater than 20 μg/dL. Low-income families, defined by PIRs, were at substantially elevated risk for having children with blood lead levels above 5 μg/dL, and the odds ratios in these families were the highest when comparing the 10-20 μg/dL group to those children with blood leads <5 μg/dL.

Among Native American children, 1 to 6 years old, living near a Superfund site in Oklahoma, strong interactions were observed between blood lead levels and poverty, suggesting that poor children were especially vulnerable to the toxic effects of lead (Malcoe et al., 2002). Moreover, blood lead levels were significantly higher in 52,407 WIC-enrolled families between 1996-2000 compared to non-WIC-enrolled families indicating the vulnerability of children with low incomes and poorer nutritional status (Zierold et al., 2004).

Similar findings have been reported from the Port Pirie prospective study (Tong et al., 2000); 375 children, 11-13 years of age, were assessed by Daniel's Scale of Occupational Prestige, which is a surrogate for SES. With adjustment for confounders, Wechsler-derived IQ scores were reported in three groups of children according to their lifetime blood lead values of <12, 12.1-17, and >17 μg/dL, respectively. For the less than 12 μg/dL group, the IQ in the high SES children was 105.6 vs. 103.1 in the low SES group. In the mid-range blood lead group

(12.1-17 µg/dL), the IQ score for the high SES group was 104.4 vs. 100.6 in the low SES group. For the higher lead group, the largest differential in IQ scores was apparent: high SES, 101.5 vs. 90.9 in the low SES group. Poor children were especially vulnerable to the neurotoxic effects of lead.

Familial and nonfamilial factors were discerned in 717 children, some of whom lived in Detroit (urban group-low SES) compared with a group of suburban-based middle class children (Breslau et al., 2001) who lived outside Detroit. Children were prospectively tested via the Wechsler at 6 and 11 years of age. Although blood lead levels were not included in this study, the results are of interest. On average, in the urban children, over time, there was a downward shift of 5 IQ points in the disadvantaged children while a negligible change was found in the suburban-middle class group. Income and demographic data for the two groups of children were defined by the 1990 U.S. Census data. Despite the absence of blood lead concentrations, it is reasonable to suggest that compared to the suburban children (according to the 1990 U. S. Census data utilized), that urban Detroit children were more likely to be exposed to lead based paint in their home environments. Familial determinants of IQ, such as maternal IQ, education, and marital status, exerted stable and uniform influences on children's IQ scores across age in both communities; none of these variables were associated with change in IQ scores. Although family factors (maternal IQ specifically) explained about two-thirds of the initial 14 point disparity in IQ scores between urban vs. suburban children, such factor(s) did not account for any part of the IQ decline of 5 points (on average). These authors concluded that IQ is a "joint" product of "genetics" and the environment. The authors emphasized that the disadvantages of inner city children, including ethnicity, housing, segregation, and educational opportunities underscores the need to fully examine extrafamilial factors, including a community's economic resources, to understand predictors of children's IQs. Although family and community factors are not completely separable, these observations reflect, in part, the legacy and vulnerability of children growing up is socioeconomically disparate communities.

Based upon the studies discussed above, there is conclusive evidence that SES has a profound influence on children's vulnerability and susceptibility to the neurotoxic effects of lead exposure.

Nutrition

There was little discussion of nutritional factors and their impacts on children's vulnerability to lead in earlier EPA documents, because very little, if any, information was available at that time. It has become evident that the dangers of lead exposure in children are substantially enhanced by diet deficient in calcium, iron, zinc and other essential nutrients; specific dietary deficiencies are not infrequently coupled to increased susceptibility to lead in low SES children.

In 205 one-year olds, who were low SES infants and who were living in old housing, blood lead levels were measured and related to nutrient intake (primarily of fat). This sample was stratified so that excessive exposure to lead could be analyzed as an independent or dependent variable to account for changes in blood lead values (Gallicchio et al., 2002). Exposure to environmental lead was assessed by measurements of lead in household dust. The authors reported a positive association between household dust levels and blood lead concentrations, and positive associations were also found between dietary intake of total calories and fat. The latter dietary observations were found to be independent of environmental exposure to lead at dust lead values above the 2001 EPA Guideline (Federal Register, 2001). These results implied that dietary control of fat and total caloric intake could have a beneficial effect on children's blood lead levels independent of environmental exposure, although the authors cautioned that control of the external environment was also a critical factor in modulating children's blood lead levels. Similar findings were reported by Lucas et al. (1996). The relationships between blood lead levels and nutritional factors were studied in 296 children who ranged in age from 9 to 72 months from low-SES families in Baltimore. When environmental lead exposure was statistically controlled, dietary fat intake had a positive association with blood lead levels, particularly in children who had blood leads >15 µg/dL.

Several studies have established a negative association between dietary iron intake (as well as biochemical assessments of iron status). Hammad et al. (1996) evaluated 299 children (9 months to 5 years of age) in a cross-sectional study design. The mean blood lead in this group was 11.4 µg/dL, and the mean age of the entire group of children was 26.2 months. After adjustment for confounders, the authors reported that the highest quartile of dietary iron intake had the lowest blood lead values. Bradman et al. (2001) studied 319 1-5 year-old children in Sacramento, CA in terms of iron status as measured by serum levels of ferritin, and

environmental exposure to lead from soil and lead-based paint. 24% of this sample was iron deficient defined as a serum ferritin level <12 ng/dL. Blood lead levels were higher for each tertile of iron-deficient children who were also experiencing excessive exposure to exogenous lead; and the greatest difference between iron-deficient compared to iron-replete children (a mean difference of 3 μg/dL) was in children who had on-going excessive exposure to lead. It was concluded that improvement of iron status, coupled to control of environmental lead sources led to a significant decline in blood lead concentrations.

An important study was carried out in an urban (Boston) primary care setting of 3,650 9-48 month-old children, and comparisons were made between iron status and "low-level" lead exposure. Iron deficiency was defined according to red blood cell indices, including mean corpuscular volume (MCV) and red cell distribution (RDW). During the study period of 1994-1996, 9.9% of the children were iron deficient, defined by cut-offs for MCV and RDW, and 9.4% of these children had blood lead concentrations of 10 μg/dL or greater (Wright et al., 1999). Among lead-poisoned children, 11.6% were iron deficient. Blood lead levels ranged from less than 5 to 44 μg/dL. More than 50% of the children screened had blood lead concentrations below 5 μg/dL; and the median blood lead was 5 μg/dL. Blood lead levels were stratified into 3 categories: less than 5 μg/dL, 5–9 μg/dL, and greater than 10 μg/dL. Chi-square analysis showed a significant association between rising blood lead levels and iron deficiency, as previously defined. In group comparisons, the mean ages of the patients with blood lead levels less than 5 μg/dL and 10 μg/dL or greater differed significantly from each other, as did those of patients with blood lead levels of less than 5 μg/dL and 5-9 μg/dL. In contrast, there was no significant difference in the mean age for patients with blood lead levels of 5-9 μg/dL and those with blood lead values of 10 μg/dL or greater. Odds ratios were calculated based upon the postulate of iron deficiency as a predictor of blood lead levels after controlling for age, hemoglobin, and insurance status. The odds ratios were 1.63 for a blood lead concentration of 5 μg/dL or more and 1.44 for a blood lead concentration of 10 μg/dL or more among iron deficient children. This study concluded that the combination of increased RDW and decreased MCV (markers of iron deficiency) is associated with blood lead concentrations of more than 5 and more than 10 μg/dL. Thus, this important study demonstrated that iron deficiency is associated with even lower blood lead levels than currently found in the United States.

However, it is important to point out that the associations is not as strong as reported in children with more severe lead poisoning, as discussed above.

In this study (Wright et al., 1999), the combined prevalence of lead poisoning and iron deficiency was present in 1.1% of the children tested. Therefore, secondary preventive measures of childhood lead poisoning, such as selective dietary interventions (iron supplementation), to reduce the intestinal absorption of lead can be simultaneously pursued in tandem with primary preventive efforts (Rosen and Mushak, 2001).

Kordas et al. (2004) examined whether iron status could account in part or for all of the negative relationship between cognitive performance and lead exposure in 602 6-8 year old children living near a metal foundry in Torreon, Mexico. The average blood lead level was 11.5 μg/dL with a standard deviation of 6.1, and 50% of this group had blood lead levels above 10 μg/dL. The results showed that the relation between blood lead and cognition was not strongly affected by nutritional status (iron and zinc), indicating that the association between blood lead and cognition was not explained by the presence of iron deficiency in a relatively intact group of children from the standpoint of nutrition (21.7% were iron deficient). However, low serum ferritin values were more prevalent in children who had blood lead concentrations above 15 μg/dL than those below 15 μg/dL (33.0 vs. 18.4%, $p<0.001$). Furthermore, successive addition of iron status did not attenuate lead's negative association with several cognitive outcomes.

Based upon these reports, it is reasonable to conclude that caloric and fat intake have important effects on blood lead levels in children. Furthermore, based on the above evidence, the inverse association between iron status and blood lead levels is clearly documented in children with low-level lead exposure to more severely elevated blood lead values.

Limited information is available concerning effects of iron supplementation on blood lead levels. Rico et al. (2005) have tested the efficacy of iron (and zinc) supplementation in 515 6-8 year-old children living in close proximity to the lead metal foundry in Torreon, Mexico. This was a randomized, double blind, placebo-controlled study with about 125 subjects in each treatment/placebo group. In addition to supplements or placebo paradigms, selective tests of cognitive functioning were also administered at baseline and 6 months later. The overall prevalence of iron and zinc deficiency was 21.7 and 28.9%, respectively. Thus, in relative terms, this was a reasonably well off population from a nutritional standpoint, with a group of children

many of whom had been excessively exposed to lead since birth. The mean blood lead level was 11.5 µg/dL (SD of 6.1). Cognitive improvements were not discerned in any of the 11 measures employed, and there was a very modest decrease in blood lead levels of only 2.6% (or about 0.30 µg/dL) for the iron supplemented group. These negative findings can be explained by the relatively intact iron status in the majority of children at baseline, and the negative results of cognitive testing can be attributed to a population of children who had been excessively exposed to lead for long timeframes. In contrast, 191 children from a community project in Costa Rica, divided into five treatment groups, had an average blood lead level of 10.98 µg/dL (Wolf et al., 2003). Oral iron supplementation led to a mean decrease in blood lead levels of 1.2 µg/dL over 3 months. These authors concluded that iron therapy can have a substantial effect on decreasing blood lead levels, particularly in children whose iron status is the most compromised and if treated promptly. Interpretation of the results of this study require some degree of caution, because of the limited sample size in each of the five groups.

Mahaffey et al. (1986) analyzed calcium intakes in comparison with blood lead levels in NHANES III. The MEAN and 25th, 50th and 75th percentiles for blood lead concentrations in 2,926 children were, respectively, 15.7, 11, 14, and 19 µg/dL, Corresponding dietary calcium intakes were, respectively, 851, 522, 789, and 1,110 mg/day. Dietary calcium intake was a significant explanatory variable for blood lead, and this relationship was inverse. Thus, in this national survey, a significant and independent inverse association was observed between dietary calcium intake, assessed by the 24-h recall method, and blood lead levels.

One hundred sixty-nine Albany-based mother-infant pairs were evaluated every 3 months during the first year of life according to calcium intakes measured by 24-h recall (Schell et al., 2004). The geometric mean value for blood leads in infants at birth was 1.6 µg/dL; this value rose to 5.1 µg/dL by 12 months of age, when 18% of the sample had elevated blood lead levels (Schell et al., 2004). A significant inverse relationship between calcium intake and blood lead values was found at 6 months; but only the inverse relationship with iron and blood leads persisted to 12 months. The majority of infants in this study met the recommended daily allowances for calcium.

Recent studies that have assessed the impact of calcium on blood lead levels have yielded reasonably consistent results. Sargent et al. (1999) studied 103 children 3.5 to 6 months of age; these infants were followed for 9 months, receiving either no treatment of calcium supplements

or treatment to increase daily calcium intake from about 450 mg/day to a supplemented intake of about 1700 mg/day. Through 4 months of supplementation, the median increase in blood leads was 57% compared to the control group. However, beyond 4 months of treatment, the effect on calcium was attenuated. Up to that time, coupled to measurements of household dust lead, calcium supplementation appeared to impair the absorption of lead from the GI tract; but this apparent effect was not sustained.

In a randomized, double blind, placebo-controlled study, 67 children (1-6 years old), whose blood lead values ranged from 10-45 μg/dL, were given a placebo or a calcium supplement to reach a daily intake of about 1800 mg/day. The mean blood lead levels at baseline were 21.4 and 20.7 μg/dL in the placebo and treatment groups, respectively. All children in this study, from an inner-city group of children in the Bronx, NY, were at or above the daily recommended allowance (RDA) for calcium. Blood lead levels declined similarly in placebo vs. treated groups over the 3-month study period without a differential in final blood lead concentrations between the two groups. It appears that there is a negative association between blood lead levels and calcium intake, particularly, in children who fall below the RDA for calcium. However, even in inner-city children, current calcium intakes appear to readily meet expected RDAs, most likely accounting for the failure of calcium supplements to have effects on blood lead values.

Genetic Polymorphisms

A paucity of information was previously available in EPA's documents in the time frame of 1986-1990. Since that time, at least three genes have been identified that may affect the accumulation and toxicokinetics of lead in children and adults. The three genes are ALAD, the vitamin D receptor gene (VDR), and the hemochromatosis gene (HFE). Relatively few studies relating to genetic polymorphisms have been reported in children compared to a substantial body of clinical research studies reported especially in excessively exposed adults. ALAD, VDR and HFE are discussed here in detail to serve as an introduction for clinical research reports in adults.

The primary importance of incorporating a discussion of genetic polymorphisms in the field of environmental health is their usefulness in detecting differences in levels of risk within specific populations (Kelada et al, 2003). The range of responses to toxic environmental exposures can vary, and population attributable risk may be substantial. Furthermore,

understanding the possible role of genetic polymorphisms in risk assessment can lead to an enhanced delineation of mechanisms underlying toxic exposures.

The ALAD gene (chromosome 9q34) encodes for ALAD, which catalyzes the second step of heme synthesis and is polymorphic. This polymorphism yields two codominant alleles, ALAD-1 and ALAD-2, and these have been differentially implicated in some clinical research studies to lead toxicity (Kelada et al., 2001). It is evident that genotypic frequencies differ by ethnicity and geography; and these considerations require careful assessment in the interpretation of research results. It has been suggested in some studies that ALAD-2 may possibly offer some level of "resistance" to the toxic effects of lead by generating a protein that avidly binds to lead, perhaps sequestering lead from its toxic expressions at various tissue sites. Other studies suggest that the rarer ALAD-2 allele has been associated with higher blood lead levels and may, thereby increase the risk of lead toxicity by producing a protein that binds more tightly than the ALAD-1 protein. Some recent studies in adults have reported that individuals homozygous for the ALAD-1 allele have higher cortical bone lead concentrations and may be at higher risk for long-term adverse effects of lead. Occupationally exposed adults have been most frequently studied in terms of the possible interaction of ALAD polymorphism and adverse health outcomes. As discussed below, reports in children concerning ALAD polymorphism and risk assessment are limited.

The vitamin D receptor (VDR) is a ligand-activated transcription factor that modulates the genomic effects of the vitamin D hormone, 1,25-dihydroxyvitamin D, in a wide variety of tissues. The gene encoding for VDR is on chromosome 12q and has common allelic variants (Zmuda et al., 2000). The allelic variants and their halotypes have been extensively studied with regard to osteoporosis susceptibility. Studies involving other disease states, such as breast and prostate cancer, diabetes, coronary artery disease, and primary hyperparathyroidism, have also focused on the role(s) of VDR gene variants. Consideration of VDR gene variants have also been extended to populations with increased lead exposure, particularly within an occupational setting. Very little information is available on these gene variants and lead exposure in the pediatric age group.

Hereditary hemochromatosis (HHC) is an autosomal recessive disorder of iron metabolism characterized by an increase in iron absorption and deposition in the liver, heart, pancreas, joints, and pituitary gland. HFE, the gene for HHC, has been mapped to the short arm

of chromosome 6 (Hanson et al., 2001). Two of the 37 allelic variants of HFE, described to date, C282Y and H63D have been significantly correlated with HHC. Homozygosity for the C282Y mutation has been found in the majority of patients and their probands diagnosed with HHC. Implications of HFE polymorphism have been proposed in studies of adults excessively exposed to lead, particularly in occupational settings. No studies of HFE have been reported in children with varying blood lead concentrations.

As yet, studies have failed to evaluate arylsulfatase (ASA) polymorphisms in lead exposed children and adults. ASA is recognized as playing a significant role in regions of the brain known to be affected by lead, and it has been established in experimental studies that lead produces low levels of ASA at sensitive stages of nervous system development (Poretz et al., 2000). Studies of ASA in children and adults may yield important information that may explain some of the neurocognitve effects of lead in pediatric and adult populations. As yet, no studies of this nature are available.

A group of 142 lead-poisoned children (mean blood lead: 27.1 μg/dL; SD: 15.2) in New York City children who expressed the 2-2 or 1-2 isozyme phenotype were reported to have blood lead levels 9-11 μg/dL higher than children who were homozygous for the ALAD-1 allele (Wetmur et al., 1991). These authors suggested the possibility that, because the ALAD-2 polypeptide binds lead more effectively, these individuals may be more susceptible to lead poisoning. At the time of publication, the lead binding properties of purified ALAD1-1 and 2-2 proteins and tissue distribution of these alleles were unknown.

The relationship was investigated between ALAD isozymes and blood lead levels in 229 Chinese children within the age range of 6-10 years old (Shen et al., 2000). The mean blood lead value was 10.3 μg/dL (SD: 3.3) and for the 92% of children homozygous for ALAD-1, the mean blood lead was 9.7 μg/dL compared with the 8% of children who were heterozygous (ALAD-1-2) and who had a mean blood lead level of 11.7 μg/dL ($p<0.05$). Using step-wise multiple regression, children who had the ALAD-2 allele were shown to be more likely to have higher blood leads compared to children who had the ALAD-1 allele.

In the only published article to date, environmental samples, blood lead levels, and nutritional factors were assessed together with determinations of VDR-Fok1 genotype (Haynes et al., 2003). A significant interaction was found between dust lead, such that at a 1 $\mu g/ft^2$ increase in floor dust lead, children with VDR-FF genotype had a 1.1% increase in blood lead;

VDR-Ff, a 0.53% increase; and VDR-ff; a 3.8% increase. At floor dust levels less than 10 μg/ft^2, children with VDR-ff had the lowest blood lead concentrations. It is noteworthy that only 17 children in this study were homozygous for the ff allele. Nonetheless, the authors suggested that VDR-Fok1 is an effect modifier for the relationship of floor dust lead exposure and blood lead concentrations.

The implications for risk assessment and health significance in these three pediatric studies are limited. Far more detailed studies in this area of investigation are needed before any firm conclusions can be reached.

Dose-Response Paradigms

The aim of this discussion is to bridge the gap between basic neurotoxicology findings assessed in Section 5.3.1 and the neurobehavioral consequences of lead discussed in Section 6.3.

Based upon current neurotoxicological studies in vivo and in vitro (Section 5.3.1) and based upon epidemiological studies of children (Section 6.3), it is biologically implausible that neurotoxic effects of lead do not occur at blood lead concentrations in children above and below 10 μg/dL. Although it is difficult to extrapolate from experimental studies to investigations in children (Manton et al., 2001), some examples are revealing.

In experimental systems, lead dose is typically employed in molar concentrations (Section 5.3.1); 10 μg/dL of lead in whole blood of children is equivalent to a molar concentration of 0.48 μM. In vitro studies have reported effects of lead on cellular regulatory systems in neurons (and other tissues) far below 0.48 μM. Whereas most or all of the lead used in neurochemistry experimental systems can participate in a reaction, only a small fraction of circulating lead in blood enters specific metabolic pathways. It is recognized that the major portion of lead in whole blood is carried by erythrocytes and that the most accessible fraction of circulating lead to other tissues is in plasma. It has been estimated that 0.24 to 0.29% of lead in whole blood is in plasma (Smith et al., 2002) and that the concentration in CSF is about half of the plasma concentration (Manton and Cook, 1984). Thus, a calculation of the latter concentrations in plasma and CSF in 10 μg/dL of whole blood yields concentrations in the low nanomolar range.

It is very unlikely that the plasma concentration of lead in the low nanomolar range is the "dose" that impacts upon the CNS of children at blood lead levels less than 10 μg/dL. The

"dose" that perturbs the central nervous system of children at "low" blood lead levels is likely to be much higher. Lead's half-lives in various tissues is a function of the site of deposition and degree of on-going exposure. In blood (absent excessive external exposure), the half-life is about 30 days and in brain, the half-life is about 2 years (Leggett, 1993). However, in the absence of on-going external exposure, blood lead levels can remain elevated for relatively extended periods of time due to mobilization from internal stores (Roberts et al., 2001; Manton et al., 2000). As a result, lead, which readily penetrates the blood-brain barrier, can continuously enter neural tissue from the blood compartment (Leggett, 1993).

Active transport mechanisms are also important to consider, and these mechanisms cause differential concentrations of lead in the systemic circulation compared to those in neuronal compartments. Metabolic pumps increase concentrations of ions within intracellular organelles to levels that exceed those in the cytosol. Mechanisms affecting Ca^{2+} distribution are the most critical (Section 5.3.1). Lead's toxic effects in the brain and other tissues are based, in large part, on its ability to "mimic" Ca^{2+} in intracellular processes, coupled to its actions to perturb the Ca^{2+} messenger system (Schanne et al., 1989; Lidsky and Schneider, 2003). For example, lead enters neurons and glia by channels that, under physiological conditions, permit the passage of Ca^{2+} (Kerper and Hinkle, 1997; Legare et al., 1998). Lead enters and damages mitochondria via cellular mechanisms that bring calcium into this organelle (Chavez et al., 1987). Thus, transport mechanisms bring about variations in local concentrations of Ca^{2+} and, presumably, lead as well (Schanne et al., 1989).

Based upon these considerations, it is concluded that brain cells in children are likely exposed to concentrations of lead, in the context of "low" blood lead levels, in the mid-nanomolar range and possibly higher, particularly in organelles that depend upon the calcium messenger system for their physiological activities. The experimental literature clearly demonstrates perturbations in fundamental cellular processes in the nanomolar range and considerably lower (Section 5.3.1).

The shape of the dose-response curve(s) of IQ and blood lead concentrations in children below blood lead levels of 10 µg/dL may be considered to be unexpected. However, the above considerations provide a different and reasonable explanation. It is accepted that lead achieves its neurotoxic effects on multi-neuronal targets, and the threshold concentrations of lead to perturb multiple CNS targets differ by orders of magnitude (Lidsky and Schneider, 2003).

For instance, second messenger systems are affected at picomolar to nanomolar concentrations (Schanne et al., 1989; Lidsky and Schneider, 2003). These perturbations can mediate a variety of toxic consequences of lead by perturbing the temporal and spatial resolution of Ca^{2+}. As a result, several loci within the complex Ca^{2+} messenger system may be impaired, thereby explaining toxic effects of lead on multiple cellular processes affecting brain functioning at differential intracellular concentrations. At somewhat higher lead concentrations in the circulation, other critical subcellular processes will be affected and impaired (e.g., heme synthesis and cellular energy metabolism). Thus, based upon various targets in the CNS affected by lead at widely different concentrations, the dose-response curve would tend to be steeper at lower lead "doses," as may be seen when inspecting the relationships between blood lead and IQ, particularly at blood lead levels less than 10 μg/dL.

Neuro-Epidemiological Studies: Implications for Individual Children

Bellinger (2004) pointed out that the clinically evident cognitive outcomes applicable to attributing specific neurobehavioral outcomes to childhood lead poisoning differ from those that are employed more typically to characterize risk in a population of children. The latter type of epidemiological studies have been applied to setting public health standards in children by the EPA and the U.S. Centers for Disease Control and Prevention. However, the clinical presentation of and ultimately the diagnosis of cognitive outcomes caused by excessive exposure to lead in individual children have received little attention. Moreover, the clinical presentation of lead exposure in the individual child cannot be clearly recognized or ascertained from epidemiological data (Lidsky and Schneider, 2005).

The majority of risk assessment studies have reported the averaged performance of large cohorts of children on a traditional IQ test as the neurobehavioral outcome measure. Such studies, after adjustment for appropriate confounders, have consistently reported an inverse correlation between blood lead concentrations and IQ scores (Schwartz, 1994). In addition to IQ as the outcome index, children who have elevated blood lead levels have been shown to lack skills in basic academic subjects (Needleman et al., 1990; Fergusson et al., 1997; Lanphear et al., 2000). As adolescents, such children are at risk for anti-social behavior (Dietrich et al., 2001; Needleman et al., 2002). These reports indicate that the outcomes in a lead-exposed child focus on impairments in intellectual achievement, academic performance, and problematic behavior.

These nonspecific outcomes are of extremely limited diagnostic utility for a pediatrician to understand what measurable outcomes may or may not be attributable to childhood lead exposure in an individual child.

Because lead has neurotoxic effects on a child's developing brain (Bressler et al., 1999; Lidsky and Schneider, 2003; Finkelstein et al., 1998), diagnostic methods are necessary to uncover manifestations of brain dysfunction. IQ tests were not designed to evaluate brain dysfunction; IQ tests are insensitive to the symptoms of brain dysfunction resulting from brain injury (Lezak, 1995). Manifestations of brain injury are manifested by highly specific aspects of impaired functions that involve language, memory, and executive skills (Lezak, 1995). In contrast, IQ is an aggregate, based on the summed performance of several sub-tests that assess an array of cognitive functions, and this array fails to tap into focal deficits that are the stigmata of brain injury. Given the lack of sensitivity of IQ scores to assess the presence of brain damage, the generally consistent findings of lead's adverse affects on IQ reflect the robustness of the reported data (Hill, 1965; Chen et al., 2005; Lanphear, 2005). Whereas the mean IQ in a large group of children has often shown a decrease as a result of brain damage, the size of the decrease fails to reflect the failures of a child's abilities to carry out daily living activities, which are typically brought to the attention of a pediatrician for treatment and management.

Neuropsychology is an applied science focused on the neurobehavioral manifestations of brain dysfunction (Lezak, 1995). Neuropsychological test batteries focus on testing paradigms that are controlled by specific neural systems to detect functional effects of brain injury. Several studies (noted above) have reported impairments in groups of children that have carried out neuropsychological tests of fine motor skills, executive abilities, language, and aspects of learning and memory (Bellinger et al., 1994; Faust and Brown, 1987; Stiles and Bellinger, 1993; Dietrich et al., 1992; Walkowiak et al., 1998; Wasserman et al., 2000; Campbell et al., 2000; Winneke and Kramer, 1997; Canfield et al., 2004; Ris et al., 2004).

Because diffuse neurocognitive "dulling" is not a typical outcome of childhood lead exposure and because a specific pattern of cognitive deficits ("signature injury or injuries") is not apparent in individual children, the clinical-pediatric presentation is specific to each individual child. Thus, a child's specific deficits evidenced by neuropsychological testing are of little assistance in making a clinical diagnosis of past or present exposure to lead and the lack of a

neurobehavioral "signature" is common to other neurotoxic agents that can cause brain injury (Hartman, 1995).

Neuropsychological testing within a clinical framework is designed to measure cognitive and behavioral manifestations of normal and abnormal brain function to arrive at a diagnosis of brain injury, when present. Decisions arriving at evidence for abnormality are based on a pattern of test results tapping specific neural systems, with the understanding that some systems will be affected and diminished as a result of brain injury whereas others will be unaffected (Lezak, 1995; Lidsky and Schneider, 2003, 2005).

From this discussion, it is reasonable to conclude that neuropsychological assessments provide additional and important information to the clinical understanding of an individual child compared to what a pediatrician and neuropsychologist can ascertain from epidemiological data. Clinical neuropsychological evaluations can lead to an etiological conclusion, together with a pediatrician's differential diagnosis, whether a child's cognitive deficits are typical of brain injury and whether, if present, that injury can be diagnostically attributed to lead exposure. When impairments are detected, it is then the task of the pediatrician to carry out a physical examination and to review medical records, radiographs, laboratory data, and environmental-exposure information. Based upon review of all this information and a differential diagnosis to rule out other causes of brain damage, a clinical determination can be made as to the etiology of an individual child's impairments and whether such deficits can be the result of lead exposure. Once alternative or contributing etiologies have been ruled out as the cause of brain damage, a diagnosis can be made causally linking lead exposure to brain damage in an individual child. This describes the collaborative roles of the neuropsychologist and pediatrician in determining the role of lead as the etiological factor (or not) in producing manifestations of brain damage in the context of different patterns of neuropsychological deficits in each individual child.

5.3.2.2 Clinical Manifestations in Adults with Childhood Lead Poisoning

It is reasonable to conclude from the studies discussed in this section that clinical manifestations become manifest in adults as persistent or latent consequences of earlier childhood lead poisoning. Specific effects of lead in this section include impairments in cognitive abilities that directly involve the central nervous system (White et al., 1993). These data have been applied to cognitive outcomes (White et al., 1993) and mortality rates in adults

following severe childhood lead poisoning (McDonald et al., 1996). Data from these analyses also indicate the presence of long-term latent and/or persistent effects on blood pressure in adults several decades after severe childhood lead poisoning (Hu, 1991). These data have been extended to more recent studies of lead's impacts on adults from early excessive childhood exposure, in terms of adverse health impacts on the central and peripheral nervous systems. With current analytical techniques, these data have been applied and connected to bone lead concentrations, as well as to the development of hypertension (Stokes et al., 1998; Gerr et al., 2002).

This section includes new concepts of health impacts of lead on adults from lead exposure during childhood, concepts that were not expressed in the previous 1986 EPA Lead AQCD/Addendum and the 1990 Supplement to that Addendum.

White et al. (1993) evaluated cognitive functioning in 33 adults (mean age of 54 years), all of whom had been admitted to Boston's Children's Hospital during 1930-1942. Because blood lead measurements were not available then, criteria for the diagnosis of lead poisoning included: (1) lead paint exposure and pica; (2) signs and symptoms of childhood lead poisoning (i.e., abdominal pain, vomiting, constipation, anorexia, irritability.) A latter subgroup of 27 adults was considered to have the mildest lead poisoning. The second and third groups had more severe central nervous system symptoms of "nerve palsy" (n = 3) and encephalopathy (n = 3) as well as (3) positive lead lines on skeletal radiographs. The 33 retrieved adults from the Boston area were generally characterized as to the severity of their childhood lead poisoning into three groups (according to the above symptoms) and according to blood lead concentrations estimated as 60–100, 90–120, and greater than 120 μg/dL, respectively. Each adult underwent a 90-m neuropsychological test battery. Compared to matched controls, the 33 adults evidenced widespread cognitive deficits in attention, memory, reasoning, motor speed, visual-spatial-constructional skills, and coordination; previously leaded subjects were lower (compared to controls) in lifetime occupational status. These observations were consistent with the onset of brain damage as children with persistence 50 years later. Exposure of the CNS during their adult years could also have occurred from release of lead from bone stores (Tsaih et al., 2001).

This is the first retrospective report that systematically addressed cognitive outcomes in adults from childhood lead poisoning. Nonetheless, these data are limited by their observational

context, lack of blood lead measurements, the long interval between childhood to the point of study as adults, the limited number of subjects, and the retrospective nature of the design.

McDonald and Potter (1996) assessed ratios of observed (O)/expected (E) deaths in a cohort of 454 adults admitted as lead-poisoned children to Boston Children's Hospital from 1923 to 1966 and traced through December, 1991. These are the only such data reported. As children, the criteria for lead poisoning was based upon the following: (1) a history of "paint pica" or other sources of exposure; (2) positive bone radiographs for lead lines; and (3) GI, neurologic and/or hematologic signs and symptoms. Seventy-six percent of this group met all three criteria and 24% met at least two out of the three criteria for diagnostic inclusion. Data were adjusted for confounders such as age, sex, ethnicity, and calendar period but not for socioeconomic status. As noted, observed deaths were compared to expected deaths; and O/E ratios were computed for hematological deaths (O/E = 9.7), for seizure disorder deaths (O/E = 5.0), for cardiovascular disease deaths (O/E = 2.1), and for cerebrovascular disease deaths (O/E = 5.5). This unique study also has limitations. It was retrospective in design, deaths may have been underestimated, there was an excess of cases dating back to the 1930s, and blood lead measurements did not begin at Boston Children's Hospital until 1963. Moreover, 153 of the original cohort of 454 were lost to follow-up. However, the authors pointed out that (1) blood lead levels measured post-1963 were generally consistent with the classification of the severity of lead poisoning pre-1963; (2) although 153 of the original group were lost to follow-up, the remaining cohort was followed for a total period of 29.5 years; (3) if deaths were missed, this would have artificially lowered the observed O/E ratios; (4) interpretation of these results could be limited by the relatively small number of deaths; but, for each of the mortality outcomes, less than one death was expected. Overall, in this cohort, mortality from all causes was about 70% higher than expected.

Collectively, although the studies reported by White et al. (1993) and McDonald and Potter (1996) have limitations, these are the first reported data to indicate that severe lead poisoning causes brain damage and impacts on mortality in adults from childhood lead poisoning.

Recent reports, utilizing current methodologies, have extended the above data relating to cognitive outcomes (Stokes et al., 1998), as well as hypertension (Gerr et al., 2002) in a cohort of 257 adults (19-29 years old) who had childhood lead poisoning at 9 months to 9 years of age

from lead smelters in Idaho's Silver Valley. In 1974-1975, the mean blood lead level in young children at each of five towns near the smelter activities was in the range of 40 to 65 µg/dL and the standard deviations of the blood lead levels in the five towns ranged from 13.5 to 28 µg/dL (Gerr et al., 2002). Of the 257 adults excessively exposed as children, 43 individual blood lead values were traced back to 1974-1975, and the mean level was 49 µg/dL. The referent cohort was in the Spokane, WA area. The exposed and nonexposed groups were compared in terms of electrophysiological and neuropsychological testing, and the latter results were evaluated with concurrent K-XRF tibial lead measurements.

Fine motor and cognitive outcomes in the exposure group, after adjustment for confounding, were significantly associated with poorer performance on hand-eye coordination, reaction time, trails B, symbol digit, serial digit learning, Raven progressive matrices, and vocabulary tests. The estimated effect of being in the exposed group was negative for all 12 of the motor and cognitive outcomes. Among tests of peripheral nerve function, vibrotactile thresholds of the fingers and standing steadiness were significantly different between the exposed and nonexposed groups; sural sensory amplitude and peroneal motor amplitude were significantly related to the exposure group. Tibial bone lead measurements failed to reach a p value $<.05$ in any of the test paradigms, although there was a trend towards significance in vocabulary and vibrotactile thresholds for fingers and toes in the exposed group. This apparent insensitivity of bone lead measurements to various outcome measures was probably related to the modest precision of K-XRF determinations.

Based on this study, it is reasonable to conclude that excessive accumulation of lead in childhood has latent and/or persistent adverse health effects on both the peripheral and central nervous systems of adults assessed 19-29 years later. The latter report, using currently available methods, is generally consistent with the earlier study by White et al. (1993). Information is needed in less severely exposed children followed longitudinally into adolescence and the adult age group.

5.3.2.3 Adults with Ambient Exposures to Lead

In the previous 1986 EPA AQCD/Addendum, the focus was on adverse health effects in adults at blood lead levels in the range of 30-50 µg/dL. The studies reviewed focused on slowed nerve conduction velocities, altered testicular function, reduced Hg production, and other signs

of impaired heme synthesis evident at somewhat lower blood lead levels. These effects pointed to a generalized impairment of normal physiological functioning as adult lead levels exceeded 30-40 µg/dL. The lowest observed effect levels of 15-30 µg/dL were related to impairments in heme synthesis. In contrast, in the 1990 Supplement to the 1986 Addendum, it was concluded that the relationship between lead and blood pressure held across a wide range of blood lead values, possibly extending down to 7 µg/dL for middle-aged men. In brief, except for effects of lead on heme synthesis down to adult blood lead values of about 15 µg/dL, EPA's emphasis was on adverse health effects in the 30-40-50 µg/dL blood lead range (1986-1990).

Since that time, studies have shown lead's effects in terms of biomarkers and indices of vulnerability and susceptibility in adult populations with blood lead concentrations, on average, less than 10 µg/dL. The number and strength of these studies are limited (see below). Several of these recent studies have also included K-XRF measurements of lead in bone which should be cautiously interpreted.

Biochemical Biomarkers

Plasma total homocysteine (tHcy) is recognized as an independent risk factor for atherosclerosis and cardiovascular disease and has both environmental and genetic risk factors. Homocysteine is an intermediate metabolite in the trans-sulfation pathway that converts methionine to cysteine. Moreover, the addition to homocysteine of serine by the pyridoxal phosphate-dependent enzyme cystathionine βeta-synthase produces cystathionine, which in turn is converted to cysteine by cystathionine gamma-lyase (CTH) (Mudd et al., 1995). Homocysteine can either be methylated back to methionine by the enzyme methionine synthase (MTH) or can undergo trans-sulfuration to produce cystathionine (Mudd et al., 1995). As a result, many enzymes can affect plasma tHcy concentrations, and each enzyme can identify a potential candidate gene for evaluating the genetic determinants of plasma tHcy. Of common gene variants, the thermolabile variant in MTHFR encoding methylenetetrahydrofolate has been associated with elevated plasma tHcy. This and other variants have been connected to the disease states noted above (Weisberg et al., 2003). In this regard, a study of 496 Caucasian adults found that common variants in CTH can be a determinant of plasma tHct levels (Wang et al., 2004).

In view of these interactions and the associations of plasma tHcy with environmental factors, cardiovascular disease and cognitive dysfunction, Schafer et al. (2005) evaluated the possible relationship between blood lead levels, tibia lead (by K-XRF), and tHcy in a longitudinal study carried out within the context of the Baltimore Memory Study. In this study, 1,140 randomly selected adults were assessed. They had a mean age of 59.3 years, an average (SD) blood lead level of 3.5 (2.4) μg/dL, and a mean (SD) tibia lead (μg of Pb/g bone mineral) concentration of 18.9 (12.5). After adjustment for age, sex, ethnicity, educational level, and tobacco and alcohol use, plasma tHcy levels were found to have increased 0.35 μmol/L per 1 μg/dL increase in blood lead concentration. No relationship was found between plasma tHcy and tibia lead levels, perhaps because of the wide standard deviation among the subjects and the modest precision of the K-XRF methodology. At blood lead levels, on average, under 10 μg/dL, these results provide some initial evidence suggesting that tHcy could be a mechanism underlying lead effects on the cardiovascular and central nervous systems. Whether lead directly elevates plasma tHcy, whether lead kinetics may be modified by tHcy, and/or whether one of homocysteine's polymorphic variants may have specific binding properties for lead are all open questions for which further investigation is required.

Vulnerability and Susceptibility

Socioeconomic Status

There was very little information on socioeconomic status (SES) in ambiently exposed adults in previous EPA Documents (1986-1990). Although some data have been published since 1990, it is limited to investigations of a female population in Mexico City (Farias et al., 1996), of male populations from the Normative Aging Study in Boston (Elreedy et al., 1999), and of a minority group of men in the Boston area (Lin et al., 2004).

Determinants of blood lead levels were evaluated in 513 pregnant women in Mexico City: one group of women was enrolled from a public general hospital, and was considered to be low SES. The second group, a high-SES cohort, was enrolled from a private hospital. The geometric mean blood lead values were 6.6 and 11.12 μg/dL from the high and low SES groups, respectively (Farias et al., 1996). The entire population of pregnant women was enrolled in this study during January 1994 to August 1995 and, beside different exposure paradigms, seasonality played an important role in differentiating blood lead concentrations between the high and low

SES groups. The primary determining factor for blood lead levels in the low SES population was the use of lead-glazed ceramics in women from the public hospital; and seasonality was the main factor influencing blood lead levels in the women from the private hospital. A predictive model, fitted to milk consumption, dietary supplements of calcium plus gestational age, was predictive of a 14 μg/dL difference between the best and worst scenarios in women from the public hospital. Seasonal differences in blood lead concentrations, which ranged, on average, from 4.7 to 12.7 μg/dL, from summer to winter, respectively, in the high SES-private hospital-based women, focused on airborne lead as their primary source of exposure, although measurements of air lead levels were not reported.

Elreedy et al. (1999) investigated various factors related to SES in 538 white males (ages 50-92) in the Normative Aging Study or Boston-based adults. Questionnaire data were collected regarding educational and occupational status, and these data were further analyzed using 1990 Census Block Group Data. Men who had four years of college, compared to others who did not graduate from high school, had, on average, lower bone lead levels. These data suggested the possibility of individual SES as having an affect on cumulative lead exposure. Detailed information on the health status of these two groups of Boston men was not provided.

Eighty-four minority individuals living in the Boston area were compared by bone lead measurements to previously studied Caucasian subjects: the mean values for blood lead (SD), tibia lead (SD) and patella lead (SD) for the minority group of males were 3.0 μg/dL, 11.9 μg/g (11), and 14.9 μg/g (15.3), respectively. These results suggest disparities in body burdens of lead in the minority group of men, particularly in those older than 55 years of age. However, the high standard deviations in the bone lead data, the modest precision of the utilized K-XRF system, and lack of information on the health status within the minority group of men require a level of caution in evaluating these outcomes, which, in themselves, based upon the NHANES data from childhood national data, are not surprising.

Nutrition

Studies reported in populations from Mexico City, Boston and Rio de Janeiro provided new information on nutritional parameters in subjects with mean blood and erythrocyte lead levels less than 10 μg/dL. These three studies examined the effects of calcium and vitamin D nutrition in various populations.

Erythrocyte lead concentrations were evaluated in 68 pregnant and 45 lactating Rio de Janeiro women whose dietary intakes of calcium were low on a chronic basis (400-600 mg/day). Whole blood lead concentrations were less than 10 µg/dL in these women, including 33 controls (Pires et al., 2001). Lactating women had significantly higher erythrocyte lead values compared to both pregnant and control subjects. Indices of bone resorption (urinary d-pyridinoline) and formation (plasma bone alkaline phosphatase) were significantly higher in pregnant and lactating women, suggesting that RBC lead was elevated in the ambiently exposed women during lactation with low dietary intakes of calcium.

A larger group of lactating women (617) in Mexico City were examined from 1994-1995 to further understand the potential effects of lowering blood lead levels through dietary calcium supplements (Hernandez-Avilla et al., 2003). The average age was 24 years; the mean blood lead level at baseline was 8.5 µg/dL. Women were randomly assigned to receive either calcium carbonate (1200 mg/day) or placebo in a double-blind study, and blood lead concentrations were measured at 3 and 6 months into the study. A modest decrease of 1.16 µg/dL (mean) was observed at 6 months in the calcium-supplemented group. This relatively small decrease in blood lead values may be explained, in part, by relatively high lead burdens in this Mexico City population, although blood lead levels did not exceed 8.5 µg/dL initially.

A cross-sectional assessment was carried out by Cheng et al. (1998) in 747 males in the context of the Boston Aging Study. In 67-year-old men (average age) the mean (SD) blood lead, tibia lead and patella lead were 6.2 (4.1) µg/dL, 21.9 (13.3) µg/g, and 32 (19.5) µg/g, respectively. After adjusting for age, education, cigarette use, and alcohol consumption, men in the lowest quintile of total dietary intakes of vitamin D (179 IU/day) had higher bone lead content compared to men in the highest quintile for vitamin D intake (IU 589/day). These data are consistent with those discussed above, in that low dietary intakes of vitamin could be expected to decrease calcium and increase lead absorption from the GI tract. However, dietary calcium intakes were not measured in this study.

Genetic Polymorphisms

Since 1986-1990, two reports have been published relating to genetic polymorphisms in ambiently exposed adults: one of these is related to ALAD and the other to HFE. Both of these studies were carried out in the Boston Normative Aging Study in adult males. Hu et al. (2001)

investigated whether ALAD polymorphism may be associated with blood and bone lead values in 726 middle-aged and elderly men from the Boston area. In this group of men, the mean (SD) of blood lead concentrations, tibia lead and patella lead were 6.2 (4.1) μg/dL, 22.1 (13.5) μg/g, and 30.4 (17.2) μg/g, respectively. The ALAD 1-1 genotype was associated with an increase of 2.55 μg/g in cortical bone (tibia), thereby suggesting the possibility that the ALAD 2 allele may decrease the accumulation of lead in bone. Whether this difference of 2.55 μg/g bone mineral was above or below the precision and/or the minimum detection limits of the K-XRF method was not addressed in this report.

Within the same Boston population, Wright et al. (2004) evaluated potential relationships between the HFE gene and bone lead values in 730 men. Of this population, 13 and 25% had the C282Y and H63D variants of HFE, respectively. After adjusting for age, smoking, and education, carriers of the HFE variant allele(s) had lower patella bone lead concentrations compared to all groups by polymorphism analyses. Caution in interpreting these data are expressed, as in other data reported from the Boston Normative Aging Study.

Neurotoxicology of Lead

One study has been reported since 1986-1990 that assessed aspects of cognitive functioning in the Normative Aging Study in Boston within a group of 466 males who had low-level or ambient lead exposure. The purpose was to evaluate whether biomarkers of lead were related to cognitive functioning, and the latter was indexed by the Mini-Mental State Examination (MMSE) (Weisskopf et al., 2004a). On two occasions, 3.5 years apart, MMSE scores were obtained during 1993–2002 in men whose age averaged 67.4 years. Bone lead measurements by K-XRF were assessed on two occasions between 1993–2002. The presented results indicated that a one-interquartile range (20 μg/g) increase in patella lead was associated with a decline in the MMSE equivalent to that of aging 5 years in relation to baseline MMSE scores. Associations were not observed in values for blood lead or tibia lead levels. The authors suggested that this steeper decline in MMSE scores was thus related to lead that is mobilizable from skeletal store (patella lead). These data reflect an important beginning to define effects of ambient lead exposure on cognitive functioning in adults. Although the MMSE has been employed in epidemiological population-based research, it is evident that a comprehensive

neuropsychological test battery has the potential to provide more definitive information related to understanding further the impact of ambient lead exposure on cognition in adults.

5.4 REPRODUCTIVE AND DEVELOPMENTAL EFFECTS OF LEAD

5.4.1 Summary of Key Findings on the Developmental and Reproductive Effects of Lead in Animals from the 1986 Lead AQCD

The 1986 Pb AQCD presented unequivocal evidence for effects of Pb on reproduction and development in laboratory animals, derived principally from studies of rodents. Fetotoxic effects (spontaneous abortion and fetal death) were reported following chronic exposures to relatively high doses (600 to 800 ppm inorganic lead) in the diet, and more subtle effects (such as changes in ALA-D activity or hematocrit) at lower doses (5 to 10 ppm in drinking water and 10 $\mu g/m^3$ in air). The 1986 Pb AQCD reported that the lowest observed adverse effect level (LOAEL) for reproductive and developmental effects was 64 µg/kg per day (multiple exposures by gavage).

The 1986 Pb AQCD also reported evidence for a variety of sublethal effects on reproduction and development in experimental laboratory animals following Pb exposure. Sublethal effects included changes in levels or function of reproductive hormones as well as effects on the gonads (both male and female) and conception. The animal data also suggested more subtle effects on hormone metabolism and reproductive cell structure. Stowe and Goyer (1971) classified the reproductive effects of Pb as gametotoxic, whether intrauterine or extrauterine.

The data reported in the 1986 Pb AQCD, and more recent studies conducted in experimental animal models, provide convincing evidence that Pb induces temporary and long-lasting effects on male and female reproductive and developmental function. The newer literature supports the earlier conclusions presented in the 1986 Pb AQCD that Pb disrupts endocrine function at multiple points along the hypothalamic-pituitary-gonad axis (Sokol et al., 1985; Stowe and Goyer, 1971; Vermande Van Eck and Meigs, 1960; Junaid et al., 1997; McGivern at al., 1991; Ronis et al., 1996, 1998b,c; Sokol, 1987; Sokol et al., 1985, 1994, 1998; Sokol and Berman, 1991; Kempinas at al., 1988, 1990, 1994; Tchernitchin et al., 1998b; Sant' Ana et al., 2001; Srivastava et al., 2004). A schematic representation of the hypothalamic-pituitary-gonadal axis is shown in Figure 5-4.1.

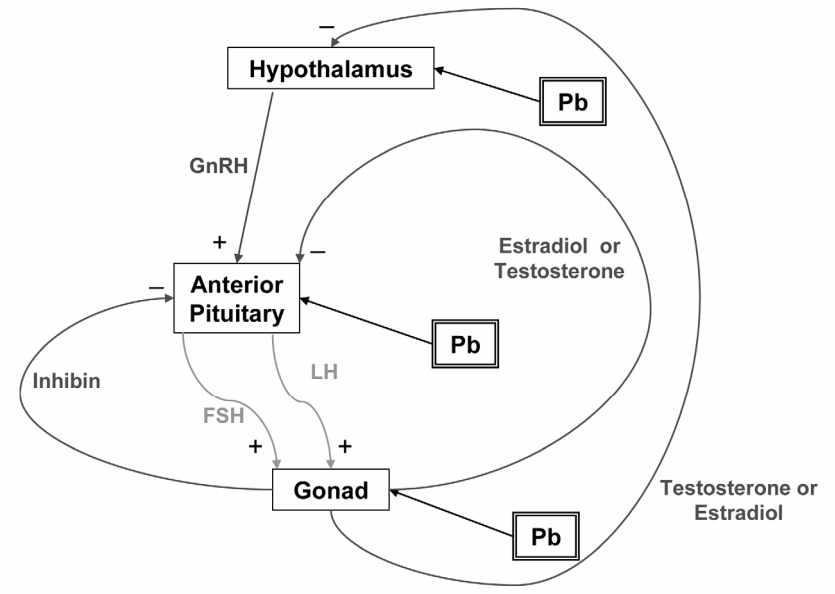

Figure 5-4.1. Data from male and female experimental animals suggests that Pb has multiple targets in the hypothalmic-pituitary-gonadal axis.

The majority of the experimental animal studies on developmental and reproductive effects of Pb examined effects due to inorganic forms of lead; very little is known about the reproductive and developmental effects due to organic forms. In general, the few available studies suggest that effects of organic forms of Pb are similar to those produced by inorganic forms. Administration of triethyl-Pb-chloride during early gestation reduces pregnancy rates in mice (Odenbro and Kihlström, 1977). Growth retardation following organolead exposure has been reported (Kennedy et al., 1975; McClain and Becker, 1972). More recent studies have demonstrated that exposure of mice to triethyl-Pb-chloride during late gestation reduces perinatal growth rate (Odenbro et al., 1988).

This section summarizes the evidence for effects of Pb exposure in developing organisms exposed during the period from conception to maturity that has been reported since 1986. Effects on neurological, immunological, or renal endpoints in developing organisms are discussed in Sections 5.3, 5.9 and 5.7, respectively.

5.4.2 Effects on Male Reproductive Function

The 1986 Pb AQCD reported convincing evidence based on experimental animal studies that Pb acts as an endocrine disruptor in males. Those studies demonstrated an association between reduced male fertility and repeat-dose Pb exposure. Lead exposure had been reported to alter sperm development and function; however, the mechanism underlying these effects was not completely understood. These effects were attributed to either alterations in testicular enzymes important for hormone production or to changes in the hormone receptors. More recent research supports the conclusion that the mechanisms for endocrine disruption in males involves Pb acting at multiple sites along the hypothalamic-pituitary-gonadal axis (see Figure 5-4.1).

Reported effects of Pb on male reproduction differ substantially across studies, with some studies finding profoundly adverse effects and other studies finding no or minimal effects. The variable findings have been attributed to the complex mechanisms involved in hormone regulation and the multiple sites of action for lead. Sokol et al. (2002) suggested that differences in results among studies may be, in part, attributed to an adaptive mechanism in the hypothalamic-pituitary-gonadal axis that may render the expression of some toxic effects dependent on exposure duration. Sokol and Berman (1991) found that timing of exposure was critical to Pb-induced male reproductive toxicity in rats. Studies conducted in nonhuman primates supported the importance of timing, finding that the adverse effects of Pb on male reproduction are dependent upon age (i.e., developmental stage at time of exposure) and duration of exposure (Foster et al., 1993; Singh et al., 1993a).

The adverse effects of Pb on male reproduction may be expressed as perturbations in sexual development and maturation, changes in fertility, endocrine disruption, and alterations in structure of reproductive cells or tissue. Each of these effects is discussed in greater detail in the sections that follow.

5.4.2.1 Effects on Male Sexual Development and Maturation

The 1986 Pb AQCD reported adverse effects of Pb on male sexual development and maturation. Experimental studies conducted in animals demonstrated that high-dose (e.g., dietary exposure to 0.08 to 1.0% Pb-acetate in mice and to 100 ppm in dogs) preadolescent Pb exposure can produce long-lasting detrimental effects on male sexual development. Numerous more recent studies conducted in experimental animals support the earlier findings that Pb

exposure during early development can delay the onset of male puberty and alter reproductive function later in life (McGivern et al., 1991; al-Hakkak et al., 1988; Chowdhuri et al., 2001; Dearth et al., 2002, 2004; Gandley et al., 1999; McGivern et al., 1991; Ronis et al., 1998a,c; Sokol et al., 1994; Yu et al., 1996). Studies that provide the strongest evidence for the dose-response range for typical effects in rodents are discussed below (Table 5-4.1).

McGivern et al. (1991) found that male rats born to dams that received Pb-acetate in drinking water beginning on gestation day 14 and through parturition (PbB 73 µg/dL) exhibited reduced sperm counts, altered male reproductive behavior, and enlarged prostates later in life. Prepubertal exposure of male Sprague-Dawley rats (age 24 to 74 days) to Pb-acetate in drinking water (PbB 30 to 60 µg/dL) resulted in significant reduction in testis weight and in the weight of secondary sex organs; however, these effects were not observed in rats exposed postpubertally (day 60 to 74; Ronis et al., 1996). A dose-dependent delay in sexual maturation was found in male rats, following prenatal Pb exposure that continued until adulthood (age 85 days) (Ronis et al., 1998a,b,c). In these studies, PbBs in the pups between the ages of 21 and 85 days were >100 µg/dL. Additional details concerning these studies are provided in Table 5-4.1.

One possible explanation for the persistent effects of Pb exposure on the male reproductive system is a disruption in pulsatile release of sex hormones during early development (Ronis et al., 1998c). Lead effects on sex hormones are discussed in Section 5.4.2.3.

5.4.2.2 Effects on Male Fertility: Effects on Sperm Production and Function

The 1986 Pb AQCD presented evidence that Pb exposure affects male fertility in various animal species, including rabbits (Cole and Bachhuber, 1915), guinea pigs (Weller, 1915), rats (Ivanova-Chemishanska et al., 1980), and mice (Schroeder and Mitchener, 1971).

Several more recent studies, conducted in various animal species, have demonstrated Pb-induced alteration of sperm parameters (e.g., count, motility, number of abnormal) (Sokol et al., 1985; and eight other studies). These effects, however, have not been reproduced in all studies. For example, Foster et al. (1996a) reported that 15- to 20-year-old cynomolgus monkeys receiving Pb-acetate for their lifetime (mean PbB 56 µg/dL) showed no significant alterations in sperm parameters (i.e., sperm count, viability, motility, and morphology) or circulating levels of testosterone (see Section 5.4.2.3 for discussion of lead-induced changes in testosterone levels).

Table 5-4.1. Selected Studies Showing the Effects of Lead on Reproductive Function in Males

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
Foster et al. (1993)	Monkey/ Cynomolgus	0–1500 µg Pb-acetate/kg-d in gelatin capsules p.o. for various durations: 9 control monkeys, 4 monkeys in lifetime group (birth to 9 years), 4 in infancy group (first 400 days of life), 4 in post-infancy exposure (from 300 days to 9 years)	Suppressed LH response to GnRH stimulation in the lifetime group ($p = 0.0370$); Sertoli cell function (reduction in the inhibin to FSH ratio) ($p = 0.0286$) in lifetime and post-infancy groups.	Lifetime group 3–26 µg/dL at 4–5 years Infancy group 5–36 µg/dL at 100–300 days, 3–3 µg/dL at 4–5 years Post-infancy group 20–35 µg/dL
Foster et al. (1996a)	Monkey/ Cynomolgus	0–1500 µg Pb-acetate/kg-d in gelatin capsules p.o. from birth until 9 years of age 8 control monkeys, 4 monkeys in low group (6–20 µg/dL), 7 monkeys in high group (22–148 µg/dL)	Mean PbB of 56 µg/dL showed no significant alterations in parameters of semen quality (count, viability, motility, or morphology).	PbB 10±3 or 56±49 µg/dL
Foster et al. (1998)	Monkey/ Cynomolgus	0–1500 µg Pb-acetate/kg-d in gelatin capsules p.o. for various durations: birth to 10 years (lifetime); PND 300 to 10 years (post-infancy); birth to 300 days (infancy); 3 control monkeys, 4 lifetime, 4 infancy, 5 post-infancy	Circulating concentrations of FSH, LH, and testosterone were not altered by treatment; semen characteristics (count, motility, morphology) were not affected by treatment possibly because not all Sertoli cells were injured; degeneration of seminiferous epithelium in infancy and lifetime groups (no difference in severity between these groups); ultrastructural alterations in seminal vesicles, most prominent in infancy and post-infancy groups.	PbB ~35 µg/dL
McGivern et al. (1991)	Rat/Sprague-Dawley	0.1% Pb-acetate in drinking water from GD 14 to parturition; 8 control litters; 6 Pb-acetate litters (5 males per litter)	Decreased sperm count (21% at 70 days and 24% at 165 days, $p<0.05$); reduced male behavior ($p < 0.05$); enlarged prostate (25% increase in weight; $p<0.07$); irregular release patterns of both FSH and LH ($p<0.05$).	Control PbB <5 µg/dL at birth Maternal PbB 73 µg/dL at birth Pup PbB 64 µg/dL at birth
Ronis et al. (1996)	Rat/Sprague-Dawley	0.6% Pb-acetate in drinking water for various durations: PND 24–74 (pubertal exposure); PND 60–74 (post pubertal exposure); 11 males and females in pubertal exposure group (10 each in control pubertal group); 6 males and females post-pubertal exposure and control groups	PbB >250 µg/dL reduced circulating testosterone levels in male rats 40–50% ($p < 0.05$); reduction in male secondary sex organ weight ($p < 0.005$); delayed vaginal opening ($p<0.0001$); disrupted estrous cycle in females (50% of rats); increased incidence of stillbirth (2% control vs. 19% Pb) ($p < 0.005$).	Pubertal PbB 30–60 µg/dL Post-pubertal PbB 30–60 µg/dL Mean PbBs in male rats 30–60 µg/dL, respectively

Table 5-4.1 (cont'd). Selected Studies Showing the Effects of Lead on Reproductive Function in Males

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
Ronis et al. (1998a)	Rat/Sprague-Dawley	0.6% Pb-acetate in drinking water ad libitum for various durations as follows: GD 5 to PND 1; GD 5 to weaning; PND 1 to weaning; 3 control litters, 2 gestation exposure litters, 2 lactation exposure litters, 2 gestation and lactation exposure litters, 2 postnatal exposure litters, 2 chronic exposure litters; 4 male and 4 female pups per litter	Suppression of adult mean serum testosterone levels was only observed in male pups exposed to Pb continuously from GD 5 throughout life ($p < 0.05$).	Group: Male PbB Naïve: 5.5±2.0 µg/dL Control: 1.9±0.2 µg/dL Gest: 9.1±0.7 µg/dL Lact: 3.3±0.4 µg/dL Gest+Lact: 16.1±2.3 µg/dL Postnatal: 226.0±29 µg/dL Chronic: 316.0±53 µg/dL
Ronis et al. (1998b)	Rat/Sprague-Dawley	Lead acetate in drinking water (0.05% to 0.45% w/v); dams exposed until weaning; exposure of pups which continued until PND 21, 35, 55, or 85; 5 control litters (0%), 10 low-dose litters (0.05%), 8 mid-dose litters (0.15%), 9 high-dose litters (0.45%); 4 male and 4 female pups per litter	Dose-response reduction in birth weight ($p < 0.05$), more pronounced in male pups; decreased growth rates in both sexes ($p < 0.05$) were accompanied by a statistically significant decrease in plasma concentrations of IGF1 through puberty PND 35 and 55 ($p < 0.05$); increase in pituitary growth hormone during puberty ($p < 0.05$).	Mean PbB in offspring at 0.05% (w/v) 49±6 µg/dL Mean PbB in offspring at 0.15% (w/v) 126 ± 16 µg/dL Mean PbB in offspring at 0.45% (w/v) 263 ± 28 µg/dL
Ronis et al. (1998c)	Rat/Sprague-Dawley	Lead acetate 0.05, 0.15, or 0.45% in drinking water beginning GD 5 continuing until PND 21, 35, 55, or 85; 5 control litters (0%), 10 low-dose litters (0.05%), 8 mid-dose litters (0.15%), 9 high-dose litters (0.45%); 4 male and 4 female pups per litter	Dose-responsive decrease in birth weight ($p < 0.05$); dose-responsive decrease in crown-to-rump length ($p < 0.05$); dose-dependent delay in sexual maturity ($p < 0.05$); decrease in prostate weight ($p < 0.05$); decrease in plasma concentration of testosterone during puberty ($p < 0.05$); decrease in plasma LH ($p < 0.05$); elevated pituitary LH content ($p < 0.05$); decrease in plasma testosterone/LH ratio at high dose ($p < 0.05$).	Dams: 0, 48, 88, or 181 µg/dL Pups PND 1: <1, 40, 83, or 120 µg/dL Pups PND 21: <1, 46, 196, or 236 µg/dL Pups PND 35: <1, 20, 70, or 278 µg/dL Pups PND 55: <1, 68, 137, or 379 µg/dL Pups PND 85: <1, 59, 129, or 214 µg/dL

Table 5-4.1 (cont'd). Selected Studies Showing the Effects of Lead on Reproductive Function in Males

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)		
Singh et al. (1993a)	Monkey/ Cynomolgus	0–1500 µg Pb-acetate/kg-d in gelatin capsules for various durations: 3 control monkeys, 4 monkeys in infancy group (exposure first 400 days), 5 in post-infancy group (exposure 300 days to 9 years of age), 4 in lifetime group (exposure from birth until 9 years)	Degeneration of seminiferous epithelium in all exposed groups (frequency not specified); ultrastructural alterations in seminal vesicles, most prominent in infancy and post-infancy groups (frequency not specified).	Chronic PbB <40–50 µg/dL		
Sokol and Berman (1991)	Rat/Wistar	0, 0.1, or 0.3% Pb-acetate in drinking water for 30 days beginning at 42, 52, or 70 days old; 8–11 control rats for each age, 8–11 rats for each age in 0.1% group, 8–11 rats for each age in 0.3% group	Dose-related suppression of spermatogenesis (decreased sperm count and sperm production rate) in the exposed rats of the two highest age groups ($p < 0.05$); dose-related suppression of serum testosterone in 52-day old rats ($p = 0.04$) and in 70-day old rats ($p < 0.003$).	Group	Age	PbB
				0%	All	<7 µg/dL
				0.1%	42 d	25 µg/dL
					52 d	35 µg/dL
					70 d	37 µg/dL
				0.3%	42 d	36 µg/dL
					52 d	60 µg/dL
					70 d	42 µg/dL

FSH, follicle stimulating hormone; GD, gestational day; GnRH, gonadotropin releasing hormone; IGF_1, insulin-like growth factor 1; LH, luteinizing hormone; PbB, blood Pb concentration; PND, post-natal day

Sokol et al. (2002) provided evidence of an adaptive mechanism in the hypothalamic-pituitary-gonadal axis in response to prolonged exposure to lead. The existence of this adaptive mechanism would explain the apparent inconsistency in reported effects on circulating testosterone levels, sperm count, and sperm production following Pb exposure. Because of this adaptive mechanism, changes in testosterone levels and certain sperm parameters may not always serve as reliable endpoints for assessing the effects of Pb on male fertility and reproductive function for all exposure durations.

Although gross changes in sperm parameters were not observed in monkeys in which chronic PbB was approximately 56 µg/dL, Foster et al. (1996a) reported that monkey sperm exhibited a statistically significant, dose-related reduction in chromatin structure (as determined by susceptibility to weak acid denaturation). These changes may have adverse impacts on fertility, and they are thought to be related to dominant lethal effects of Pb (similar to the effects reported by al-Hakkak et al. [1988] in mice). Additional details concerning Foster et al. (1996a) are provided in Table 5-4.1.

The data from Foster et al. (1996a), demonstrating a change in monkey sperm chromatin suggestive of a subtle lead-induced reduction in male fertility (in the absence of gross changes sperm parameters), are consistent with observations of reduced in vitro fertilization capacity of sperm collected from other mammalian species. Sokol et al. (1994) reported that exposure of adult male rats to Pb-acetate in drinking water for 14 to 60 days (PbB 33 to 46 µg/dL) resulted in reduced in vitro fertilization of eggs harvested from unexposed females. No differences were observed in sperm ultrastructure or in the DNA histogram of sperm obtained from lead-exposed rats compared to controls. Consistent with this finding are reports of reduced fertilization capacity of rabbit sperm exposed to high concentrations (25 µM) of Pb chloride in vitro (Foote, 1999) and reduced in vitro fertilization capacity of sperm from mice exposed to Pb in drinking water at 1 g/L for 4 months (PbB not reported) (Johansson et al., 1987).

Two modes of action have been proposed for lead-induced alterations in sperm capacity for fertilization. The affinity of Pb for sulfhydryl groups may explain some of the lead-induced alterations in sperm structure and function. Mammalian sperm possess high concentrations of sulfhydryl groups, which are critical for maintaining normal function (Johansson and Pellicciari, 1988). Reyes et al. (1976) demonstrated that binding of Pb to membrane thiols inhibits sperm maturation. In addition, recent experimental data also suggest that lead-induced generation of

reactive oxygen species (ROS) may contribute to the injury of tissues responsible for sperm formation (see Section 5.4.2.4).

5.4.2.3 Effects on Male Sex Endocrine System

The 1986 Pb AQCD reported that, although the mode of action for the adverse effects of Pb on the male reproductive system was not understood, effects on hormone production or hormone receptors were likely contributors. More recent studies provide convincing evidence that Pb acts as an endocrine disruptor in males at various points along the hypothalamic-pituitary-gonadal axis (Figure 5-4.1). In rats, Pb exposures that decreased serum testosterone levels increased mRNA levels of GnRH and LH in the hypothalamus and pituitary, respectively, and increased levels of LH in pituitary; these changes can occur in the absence of a change in serum gonadotropin levels (Klein et al., 1994; Ronis et al., 1998c; Sokol et al., 2002). In monkeys, chronic Pb exposures (PbB 20 to 35 μg/dL) suppressed GnRH-induced secretion of LH and decreased serum testosterone:LH and inhibin:FSH ratios (Foster et al., 1993). The mechanisms underlying the effects on the hypothalamic-pituitary-gonadal axis have not been elucidated but may involve a suppression of GnRH secretion (Bratton et al., 1994; Sokol, 1987; Sokol et al., 1998).

Although there is evidence for a common mode of action, consistent effects on circulating testosterone levels are not always observed in lead-exposed animals. Rodamilans et al. (1988) and Kempinas et al. (1994) attributed these inconsistencies to the normal biological variation (circannual and seasonal) of testosterone secretion in rats and monkeys. Observations of lead-induced reductions in testosterone levels in some studies, but not others, may be due to enhanced sensitivity to inhibition of the testosterone secretory system during certain periods of development. In addition, the hypothalamic-pituitary-gonadal axis exhibits compensatory mechanisms that may attenuate the effects of Pb during prolonged Pb exposure (Sokol et al., 2002). Taken together, the sensitivity of testosterone secretion during certain periods and potential for modulation of the effects during long-term exposures studies, may explain some of the apparent inconsistencies in the reported effects of Pb exposure on circulating testosterone levels.

5.4.2.4 Effects on Morphology and Histology of Male Sex Organs

The 1986 Pb AQCD reported evidence for histological changes in the testes or prostate in rats, in association with relatively high doses of Pb (Chowdhury et al., 1984; Hilderbrand et al., 1973; Golubovich et al., 1968). More recent studies conducted in animal models provide persuasive support for testicular damage (i.e., ultrastructural changes in testes and cytotoxicity in Sertoli cells) following lower level lead exposure (Foster et al., 1998; Singh et al., 1993a; Batra et al., 2001; Chowdhury et al., 1986, 1987; Corpas et al., 1995; Pinon-Lataillade et al., 1993; Saxena et al., 1990). Studies conducted in nonhuman primates warrant particular attention. These studies found ultrastructural changes in the testes (Sertoli and other spermatogenic cells) of monkeys at PbB 35 to 40 µg/dL (Foster et al., 1998; Singh et al., 1993a).

Foster et al. (1998) reported that chronic Pb exposure (PbB ~35 µg/dL), beginning in infancy, resulted in persistent ultrastructural changes in the testes of cynomolgus monkeys. Electron microscopy showed disruption of the general structure of the seminiferous epithelium involving Sertoli cells, basal lamina, and spermatids in the groups exposed for lifetime and during infancy (with no duration difference in severity). Chronic exposures to Pb beginning after infancy, that achieved similar PbBs, did not produce these effects.

Similarly, Singh et al. (1993a) demonstrated ultrastructural changes in testicular basement membrane and Sertoli cell morphology (seminiferous tubules) in cynomolgus monkeys exposed chronically to Pb (PbB <40 to 50 µg/dL); the effects were most prominent when dosing began in infancy or post-infancy. These results suggest that, in monkeys, Pb exposure during certain periods of development produces persistent testicular alterations. Additional details concerning Foster et al. (1998) and Singh et al. (1993a) are provided in Table 5-4.1.

A possible mode of action for lead-induced testicular injury is oxidative stress. Foster et al. (1998) suggested that lead-induced oxygen free radical generation was a plausible mechanism of testicular injury in primates. This oxygen radical hypothesis is supported by studies conducted in rodents (Chowdhury et al., 1984; Acharya et al., 2003; Adhikari et al., 2001; Batra et al., 2001; Bizarro et al., 2003; Chowdhury et al., 1984; Gorbel et al., 2002; Mishra and Acharya, 2004). Also supporting the oxidative stress hypothesis are observations of increases in the percentage of apoptotic cells in the testes of rodents in response to Pb exposure (Pace et al., 2005; Gorbel et al., 2002; Adhikari et al., 2001).

5.4.3 Effects on Female Reproductive Function

Lead has been shown to disrupt the hypothalamic-pituitary-gonadal axis and to produce ovarian atrophy and reproductive dysfunction in females (Figure 5-4.1). The 1986 Pb AQCD reported that Pb exposure was associated with inhibition of menstruation, ovulation, and follicular growth in monkeys (Vermande-Van Eck and Meigs, 1960), and in rodents Pb exposure delayed vaginal opening, decreased frequency of implantation, and reduced rates of pregnancy (Kimmel et al., 1980; Odenbro and Kihlström, 1977, respectively).

Data from more recent experimental animal studies support these findings. Lead effects on female reproduction may be classified as alterations in female sexual maturation, effects on fertility and menstrual cycle, endocrine disruption, and changes in morphology or histology or female reproductive organs as well as the placenta. Recent literature concerning each of these effects is summarized below.

5.4.3.1 Effects on Female Sexual Development and Maturation

The 1986 Pb AQCD reported that Pb exposure in rodents produced delays in sexual maturation. Grant et al. (1980) reported delayed vaginal opening in female rats exposed in utero and during lactation and maturation (PbB ~20 to 40 μg/dL). More recent studies in experimental animals (primarily rodent studies) provide convincing evidence that Pb exposure before puberty (particularly prenatal and early postnatal exposure) delays the maturation of the female reproductive system (Dearth et al., 2002, 2004; Ronis et al., 1996, 1998b,c).

Dearth et al. (2002) is of particular interest, because it employed a cross-fostering design (to allow comparison of pups exposed during gestation only, lactation only, or both) and because maternal and offspring PbBs were monitored throughout gestation and lactation. Fisher 344 dams were exposed to Pb by gavage beginning 30 days before mating until weaning of the pups at 21 days of age (gavage exposure removes possible confounding of exposure by consumption of Pb in drinking water by pups in those studies where drinking water is the route of exposure for dams). Mean maternal PbB was approximately 40 μg/dL. Pups exposed during gestation and lactation had the highest PbB (38.5 μg/dL) on day 10; at this time, the PbBs in pups exposed during gestation only or lactation only were 13.7 and 27.6 μg/dL, respectively. By postnatal day (PND) 30, all three groups had PbB ≤3 μg/dL. Dearth et al. (2002) reported a statistically significant delay in the onset of puberty (vaginal opening and days at first diestrus) in rats

exposed during lactation, gestation, or during lactation and gestation (with no differences among the groups). In addition, a statistically significant reduction in the circulating levels of insulin-like growth factor 1 (IGF_1), LH, and estradiol (E_2) were reported on PND 30 in all three treatment groups (with no differences among treatment groups). Additional details concerning Dearth et al. (2002) are provided in Table 5-4.2.

A subsequent study in both Sprague-Dawley and F344 rats (Dearth et al., 2004) showed that the F344 strain is more sensitive to maternal Pb exposure than Sprague-Dawley rats to lead-induced delayed puberty, which could, in part, explain the inconsistencies with effect levels observed in Sprague Dawley rats (e.g., Ronis et al., 1998a,b,c; McGivern et al., 1991). Ronis et al. (1998c) suggested that the delayed onset of puberty may arise from a lead-induced disruption of pulsatile release of sex hormones (see Section 5.4.3.3).

5.4.3.2 Effects on Female Fertility

The 1986 Pb AQCD reported convincing evidence from experimental animal studies for lead-induced alterations in female fertility, including interference with implantation and pregnancy (Odenbro and Kihlström, 1977; Wide and Nilsson, 1977). More recent studies have confirmed these effects. In general, Pb exposure does not produce total sterility, although Pb exposure clearly disturbs female fertility (Taupeau et al., 2001). Studies in nonhuman primates and rodents have shown that exposure of gravid females to Pb produces implantation dysfunction and reduces litter size and newborn survival (Lögdberg et al., 1987; Flora and Tandon, 1987; Johansson and Wide, 1986; Pinon-Lataillade et al., 1995; Piasek and Kostial, 1991; Ronis et al., 1996). See Section 5.4.4.1 for details.

5.4.3.3 Effects on the Female Sex Endocrine System and Menstrual Cycle

The 1986 Pb AQCD described numerous studies that found effects of Pb on the female endocrine system and menstrual cycle in various species, including nonhuman primates, and that supported the conclusion that Pb was an endocrine disruptor in females (Grant et al., 1980; Maker et al., 1975; Vermande-Van Eck and Meigs, 1960). Observations of delayed vaginal opening (see Section 5.4.3.1) were attributed to the endocrine disruption effects of Pb on the hypothalamic-pituitary-gonadal axis (Stowe and Goyer, 1971; Vermande Van Eck and Meigs, 1960).

Table 5-4.2. Selected Studies Showing the Effects of Lead on Reproductive Function in Females

Citation	Species/ Strain	Dose/Route/Form/Duration/ Group Size	Endpoint/Magnitude of Effect (% or incidence) /p-value	Blood Lead Concentration (PbB)
Dearth et al. (2002)	Rat/Fisher 344	12 mg/mL Pb-acetate gavage from 30 days prior breeding until pups were weaned 21 day after birth; 10–32 litters per group, control group, gestation and lactation exposure, gestation only exposure, lactation only exposure	Delay in onset of puberty ($p < 0.05$); reduced serum levels of IGF_1 ($p < 0.001$), LH ($p < 0.001$), and E_2 ($p < 0.001$).	Maternal PbB ~40 µg/dL Pups PbB as follows: Gest+lact ~38 µg/dL PND 10 Gest+lact ~15 µg/dL PND 21 Gest+lact ~3 µg/dL PND 30 Gest ~14 µg/dL PND 10 Gest ~3 µg/dL PND 21 Gest ~1 µg/dL PND 30 Lact ~28 µg/dL PND 10 Lact ~15 µg/dL PND 21 Lact ~3 µg/dL PND 30
Foster (1992)	Monkey/ Cynomolgus	Daily dosing for up to 10 years with gelatin capsules containing Pb-acetate (1.5 mg/kg); 8 control group monkeys, 8 lifetime exposure (birth–10 years), 8 childhood exposure (birth–400 days), and 8 adolescent exposure (PND 300-10 years of age)	Statistically significant reductions in circulating levels of LH, ($p < 0.042$), FSH ($p < 0.041$), and E_2 ($p < 0.0001$) during menstrual cycle; progesterone concentrations were unchanged and menstrual cycle was not significantly affected.	PbB <40 µg/dL
Foster et al. (1992)	Monkey/ Cynomolgus	Daily dosing for up to 10 years with gelatin capsules containing Pb-acetate (1.5 mg/kg); 8 control group monkeys, 8 childhood (birth–400 days), 7 adolescent (PND 300–10 years), 7 lifetime (birth–10 years)	No effect on endometrial response to gonadal steroids as determined by ultrasound.	PbB <40 µg/dL
Foster et al. (1996b)	Monkey/ Cynomolgus	Chronic exposure to Pb-acetate 50 to 2000 µg/kg-day p.o. beginning at birth for 15–20 years; 20 control monkeys, 4 monkeys in 50 µg/kg-d group, 3 monkeys in 100 µg/kg-d, 2 monkeys in 500 µg/kg-d group, and 3 monkeys in 2000 µg/kg-d group	Reduced corpora luteal production of progesterone ($p = 0.04$), without alterations in E_2, 20-alpha-hydroxyprogesterone, or menstrual cyclicity.	PbB 10–15 µg/dL in low group (50 or 100 µg/kg-day) PbB 25–30 µg/dL in moderate group (500 or 2000 µg/kg-day)

Table 5-4.2 (cont'd). Selected Studies Showing the Effects of Lead on Reproductive Function in Females

Citation	Species/ Strain	Dose/Route/Form/Duration/ Group Size	Endpoint/Magnitude of Effect (% or incidence)/p-value	Blood Lead Concentration (PbB)
Franks et al. (1989)	Monkey/ Rhesus	Lead acetate in drinking water (2–8 mg/kg-d) for 33 months; 7 control and 10 Pb monkeys	Reduced circulating concentration of progesterone ($p < 0.05$); treatment with Pb did not prevent ovulation, but produced longer and more variable menstrual cycles and shorter menstrual flow.	PbB 68.9 ± 6.54 µg/dL
Laughlin et al. (1987)	Monkey/ Rhesus	Lead acetate in drinking water at 3.6, 5.9, or 8.1 mg/kg-day for 1–2 years 7 control and 10 experimental monkeys per group	Reductions in cycle frequency ($p<0.01$); fewer days of flow ($p<0.01$); longer and more variable cycle intervals ($p<0.025$).	PbB 44–89 µg/dL 51.2 µg/dL (low dose) 80.7 µg/dL (mid dose) 88.4 µg/dL (high dose)
Lögdberg et al. (1988)	Monkey/ Squirrel	Lead acetate (varying concentrations ≤0.1% in diet) maternal dosing from 5-8.5 weeks pregnant to PND 1 11 control monkeys, 3 low Pb exposure group (PbB 24 µg/dL), 7 medium Pb group (PbB 40 µg/dL, 5 high Pb group (PbB 56 µg/dL)	Dose-dependent reduction in placental weight ($p<0.0007$); various pathological lesions were seen in the placentas ($n = 4$), including hemorrhages, hyalinization of the parenchyma with destruction of the villi and massive vacuolization of chorion epithelium.	Mean maternal PbB 37 µg/dL (22-82 µg/dL) 24 (22–26) µg/dL (low dose) 40 (35–46) µg/dL (mid dose) 56 (43–82) µg/dL (high dose)

E_2, estradiol; FSH, follicle stimulating hormone; GD, gestational day; IGF_1, insulin-like growth factor 1; LH, luteinizing hormone; PbB, blood Pb concentration; PND, post-natal day

More recent studies have provided convincing support for endocrine-mediated alterations of the female reproductive system in rats (Srivastava et al., 2004; Dearth et al., 2002; Ronis et al., 1998a,b,c; Junaid et al., 1997; Ronis et al., 1996), guinea pigs (Sierra and Tiffany-Castiglioni, 1992), and nonhuman primates (Foster et al., 1992, 1996b; Foster, 1992; Franks et al., 1989; Laughlin et al., 1987). The nonhuman primate studies are particularly relevant to extrapolations to humans and provide dose-response information for effects of Pb on female sex hormones and menstrual cycle.

Laughlin et al. (1987) found that exposure to Pb (PbB 44 to 89 μg/dL) alters menstrual cycles (specifically, causing reductions in cycle frequency, fewer days of menstrual flow, and longer and more variable cycle intervals) in female rhesus monkeys. Consistent with these observations, Franks et al. (1989) found that chronic exposure to Pb in the drinking water (PbB 70 μg/dL) reduced circulating concentrations of progesterone (suggesting impaired luteal function), produced longer and more variable menstrual cycles and temporally shorter menstrual flow in female rhesus monkeys. Additional details concerning these studies are provided in Table 5-4.2.

At lower blood Pb levels (PbB <40 μg/dL), female cynomolgus monkeys exhibited statistically significant reductions in circulating levels of LH, FSH, and E_2 during the menstrual cycle; however, serum progesterone concentrations were unchanged and menstrual cycle was not significantly affected (Foster, 1992). Similar exposures and PbB were shown to have no effect on endometrial response to gonadal steroids in cynomolgus monkeys as determined by ultrasound analysis (Foster et al., 1992). At lower blood lead concentrations (25 to 30 μg/dL), reduced corpora luteal production of progesterone occurred in the absence of alterations in E_2, 20-alpha-hydroxyprogesterone, or menstrual cyclicity (Foster et al., 1996b). In contrast to Foster et al. (1992), this study (Foster et al., 1996b) found no statistically significant effect of Pb on serum progesterone levels in cynomolgus monkeys that had lower PbB (10 to 15 μg/dL). Additional details concerning these studies are provided in Table 5-4.2.

Several modes of action for lead-induced, endocrine disruption-mediated alterations in female reproduction have been proposed, including changes in hormone synthesis or metabolism at the enzyme level (Wiebe and Barr, 1988; Wiebe et al., 1988) and changes in hormone receptor levels (Wiebe et al., 1988; Wide and D'Argy, 1986). In addition, Pb may alter sex hormone release and imprinting during early development (Ronis et al., 1998c; Tchernitchin et al.,

1998a,b). The latter effects would be consistent with observations of persistent changes in estrogen receptor levels in the uterus (Wiebe and Barr, 1988) and LH function in the ovary (Srivastava et al., 2004) in lead-exposed animals.

5.4.3.4 Effects on Morphology and Histology of Female Sex Organs and the Placenta

Lead-induced changes in morphology or histology in female sex organs and the placenta may explain reduced fertility and impaired female reproductive success (see Sections 5.4.3.2 and 5.4.4.1.). Lögdberg et al. (1988) reported a dose-dependent reduction in placental weight and an increase in pathological lesions of the placenta in squirrel monkeys that received oral doses of Pb-acetate (0.001 to 0.1% in diet) during the last three-fourths or two-thirds of pregnancy (mean maternal PbB 37 µg/dL; range: 22 to 82 µg/dL). These effects occurred without overt toxicity in the mothers. Additional details concerning Lögdberg et al. (1988) are provided in Table 5-4.2.

Similar effects on placental weight and histology were observed in mice (Fuentes et al., 1996; Nayak et al., 1989). These effects on the placenta may explain the reduced birth weight that has been associated with prenatal Pb exposure (see Section 5.4.5). Exposure to Pb in early pregnancy also produces structural changes in the epithelium of the uterus of mice (Nilsson et al., 1991; Wide and Nilsson, 1979). These changes in uterine tissue may impair successful implantation of the blastocysts (see Section 5.4.4.1).

5.4.4 Effects on Embryogenesis

Lead exposure can increase fetal mortality, produce a variety of sublethal effects, and disrupt the growth and development of the offspring. Many of the lead-induced sublethal developmental effects occur at maternal PbB levels that do not result in clinical toxicity in the mothers.

5.4.4.1 Embryo/Fetal Mortality

The 1986 Pb AQCD concluded that that acute exposure to high doses of Pb interfered with implantation and pregnancy (Wide, 1985; Odenbro and Kihlström, 1977; Wide and Nilsson, 1977; Vermande-Van Eck and Meigs, 1960). This conclusion is supported by results of more recent studies (Lögdberg et al., 1987; Giavini et al., 1980; Jacquet, 1976, 1977; Jacquet et al.,

1975, 1976; Johansson and Wide 1986; Johansson et al., 1987; Johansson, 1989; Maisin et al., 1978; Pinon-Lataillade et al., 1995; Wide and Nilsson, 1977, 1979).

Lögdberg et al. (1987) reported an increase in pre- and perinatal mortality in squirrel monkeys that received Pb-acetate orally during the last two-thirds of pregnancy (45% versus 7 to 8% among controls). Mean maternal PbB was 54 μg/dL (39 to 82 μg/dL). These fetotoxic effects occurred without overt toxicity in the mothers. Additional details concerning Lögdberg et al. (1987) are provided in Table 5-4.3. These effects are consistent with data from rodent studies, wherein gestational exposure to Pb (PbB 32 to >70 μg/dL) resulted in smaller litters and fewer implantation sites (e.g., Pinon-Lataillade et al., 1995; Singh et al., 1993b; Piasek and Kostial, 1991).

Numerous studies have been performed to elucidate the mechanisms by which Pb causes prenatal death (Maisin at al., 1978; Jacquet, 1977, 1976; Jacquet et al., 1976, 1975). The available data suggest that Pb may alter blastocyst development and impair implantation. Hanna et al. (1997) demonstrated that in vitro exposure of 2- and 4-cell mouse embryos to 200 μM Pb-acetate resulted in reduced cell proliferation and blastocyst formation. Additional evidence for an effect on blastocysts is provided by data from in vitro fertilization studies (Chowdhuri et al., 2001; Johansson, 1989; Johansson et al., 1987). Johansson and co-workers (1989, 1987) reported that Pb delayed the timing of escape from the zona pellucida and induced a premature acrosome reaction. These effects could disrupt attachment and implantation of the blastocyst if they were to occur in vivo.

5.4.4.2 Effects on embryo/fetal morphology

The 1986 Pb AQCD summarized numerous reports that found associations between prenatal exposure to high doses of Pb and increased incidences of teratogenic effects (particularly tail stunting) in rodents (Ferm and Carpenter, 1967; Dey et al., 2001; Flora and Tandon, 1987; Ronis et al., 1996; Wide, 1985). More recent studies provide additional support for teratogenic effects of Pb in experimental animals (Flora and Tandon, 1987). Flora and Tandon (1987) demonstrated a dose-dependent effect on the incidence of tail malformations at ≥10 mg/kg i.v. on days 9 to 11 of gestation (PbB 13 to 45 μg/dL) that occurred only in those dams exhibiting "observable maternal toxicity" (not otherwise specified in the report). The few

Table 5-4.3. Selected Studies Showing the Effects of Lead on Mammalian Embryogenesis and Development

Citation	Species/ Strain	Dose/Route/Form/Duration/Group size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
Cory-Slechta et al. (2004)	Rat/Long-Evans	Lead acetate in drinking water (150 ppm); 2 months before breeding until the end of lactation; 14 rats no maternal stress with Pb exposure, 15 rats no maternal stress with Pb exposure, 18 rats maternal stress without Pb exposure, 23 rats maternal stress and Pb exposure	Pb alone (in male) ($p<0.05$) and Pb plus stress (in females) ($p<0.05$) permanently elevated corticosterene levels in offspring	PbB 30–40 µg/dL
Dearth et al. (2002)	Rat/Fisher 344	12 mg/mL Pb-acetate gavage during gestation and lactation exposure 4 groups: control group, gestation and lactation exposure, gestation only exposure, lactation only exposure 10–32 litters per group (NOS)	Delayed onset of puberty ($p<0.05$); suppressed serum levels of IGF_1, LH, and E_2 ($p<0.001$); Pb altered translation and/or secretion of IGF_1 ($p<0.001$).	Maternal PbB ~40 µg/dL Pups PbB as follows: Gest+lact ~38 µg/dL PND 10 Gest+lact ~15 µg/dL PND 21 Gest+lact ~3 µg/ PND 30 Gest ~14 µg/dL PND 10 Gest ~3 µg/dL PND 21 Gest ~1 µg/dL PND 30 Lact ~28 µg/dL PND 10 Lact ~15 µg/dL PND 21 Lact ~3 µg/dL PND 30
Flora and Tandon (1987)	Rat/Albino (NOS)	Lead nitrate dissolved in water 2–20 mg/kg-d i.v. on day 9, 10, 11 of gestation; 6 rats in each group (0, 5, 10 , 20, 40 mg/kg lead)	Dose-dependant increase in external malformations at all doses ($p<0.001$), particularly tail defects; dose-dependant decrease in number of live births at 20 and 400 mg/kg ($p<0.001$); dose-dependent increase in number of resorptions per dam at ≤10 mg/kg ($p<0.01$).	PbB 4.13±0.61 µg/dL 0 mg/kg PbB 10.21±0.61 µg/dL 5 mg/kg PbB 13.13±0.27 µg/dL 10 mg/kg PbB 29.41±0.41 µg/dL 20 mg/kg PbB 45.03±0.31 µg/dL 40 mg/kg
Fox et al. (1991a)	Rat/Long-Evans hooded	Lactation exposure via dams exposed to 0.02 or 0.2% Pb in drinking water from PND 1 through weaning (PND 21); 8 female pups per litter (number of litter unspecified) control pups, 8 pups for litter (number of litter unspecified) low-level exposure pups, 8 pups per litter (number of litter unspecified) moderate level exposure pups	Long-term, dose-dependent decreases retinal Na/K ATPase activity in the female offspring (only female pups were used) (-11%; -26%) ($p<0.05$).	PbB 18.8 µg/dL (0.02%) or 59.4 µg/dL (0.2%) at weaning

Table 5-4.3 (cont'd). Selected Studies Showing the Effects of Lead on Mammalian Embryogenesis and Development

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
Fox et al. (1997)	Rat/Long-Evans hooded	0.02 or 0.2% Pb-acetate in drinking water from PND 0–PND 21; 8 female pups per litter control pups; 8 pups per litter low-level exposure; 8 pups per litter moderate level exposure (number of litters per dose unspecified)	Developmental and adult Pb exposure for 6 weeks produced age and dose-dependent retinal degeneration such that rods and bipolar cells were selectively lost; at the ultrastructural level, all dying cells exhibit the classical morphological features of apoptotic cell death; decrease in the number of rods was correlated with the loss of rhodopsin content per eye confirming that rods were directly affected by Pb ($p < 0.05$); single-flash rod ERGs and cone ERGs obtained from lead-exposed rats demonstrated that there were age- and dose-dependent decreases in the rod a-wave and b-wave sensitivity and maximum amplitudes without any effect on cones; in adult rats exposed to Pb for three weeks, qualitatively similar ERG changes occurred in the absence of cell loss or decrease in rhodopsin content ($p < 0.05$); developmental and adult Pb exposure for three and six weeks produced age- and dose-dependent decreases in retinal cGMP phosphodiesterase (PDE) activity resulting in increased cGMP levels ($p<0.05$); retinas of developing and adult rats exposed to Pb exhibit qualitatively similar rod mediated ERG alterations as well as rod and bipolar apoptotic cell death ($p<0.05$) Similar biochemical mechanism such as the inhibition of rod and bipolar cell cGMP PDE, varying only in degree and duration, underlies both the lead-induced ERG rod-mediated deficits and the rod and bipolar apoptotic cell death ($p<0.05$).	PbB weanlings 19±3 (low exposure) or 59±8 µg/dL (moderate exposure), adult 7±2 µg/dL (at PND 90)
Iavicoli et al. (2003)	Mouse/Swiss	Lead acetate in food (0.02, 0.06, 0.11, 0.2, 2, 4, 20, 40 ppm); exposure began 1 day after mating until litter was 90 days old; one litter of mice exposed to each dietary concentration	Low-level Pb exposure (PbB 2–13 µg/dL) reduced red cell synthesis ($p<0.05$); high-level exposure (PbB 0.6-2 µg/dL) enhanced red cell synthesis ($p < 0.05$).	PbB 0.6 to <2.0 µg/dL or >2.0–13 µg/dL

Table 5-4.3 (cont'd). Selected Studies Showing the Effects of Lead on Mammalian Embryogenesis and Development

Citation	Species/ Strain	Dose/Route/Form/Duration/Group size	Endpoint/Magnitude of effect/p-value	Blood Lead Concentration (PbB)
Lögdberg et al. (1987)	Monkey/ Squirrel	Lead acetate (5–20 mg/kg daily to maintain PbB) maternal dosing from 5–8.5 weeks pregnant to PND1 20 control; 11 lead-exposed monkeys	Increase in pre- and perinatal mortality among squirrel monkeys receiving Pb-acetate p.o. during the last two-thirds of pregnancy (45% vs. 7–8% among controls). Statistically significant reductions in mean birth weight ($p<0.05$) were observed in Pb exposed monkeys as compared to controls. Effects occurred without clinical manifestation of toxic effects in the mothers.	PbB 54 µg/dL (39–82 µg/dL)
Ronis et al. (1996)	Rat/Sprague-Dawley	0.6% Pb-acetate in drinking water for various durations: PND 24–74 (pubertal exposure); PND 60–74 (post pubertal exposure); 11 males and females in pubertal exposure group (10 each in control pubertal group) 6 males and females post-pubertal exposure and control groups	Reduction in serum testosterone levels in male, not female; in female suppression of circulating E2 ($p<0.05$) and LH ($p<0.05$); reduction in male secondary sex organ weight ($p<0.0005$); delayed vaginal opening and disrupted diestrous in females ($p<0.005$); increased incidence of stillbirth (2% control vs. 19% Pb) ($p<0.005$).	*In utero* PbB 250–300 µg/dL pre-pubertal PbB 30–60 µg/dL post pubertal PbB 30–60 µg/dL PbBs in the dams and offspring in this experiment were >200 µg/dL
Ronis et al. (1998a)	Rat/Sprague-Dawley	0.6% Pb-acetate in drinking water *ad libitum* for various durations; GD 5 to PND 1; GD 5 to weaning; PND 1 to weaning; 3 control litters, 2 gestation exposure litters, 2 lactation exposure litters, 2 gestation and lactation exposure litters, 2 postnatal litters, 2 chronic litters (4 male and 4 female pups per litter)	Dose-dependent delay in sexual maturation (delayed vaginal opening) ($p<0.0002$) following prenatal Pb exposure that continued until adulthood (85 days old); reduced birth weight ($p<0.05$), more pronounced among male pups.	Group / Pup PbB Naïve ~6 µg/dL Control <2 µg/dL Gest ~10 µg/dL Lac ~3 µg/dL Gest+Lac ~13 µg/dL Postnatal ~260 µg/dL Chronic ~287 µg/dL

Table 5-4.3 (cont'd). Selected Studies Showing the Effects of Lead on Mammalian Embryogenesis and Development

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
Ronis et al. (1998b)	Rat/Sprague-Dawley	Lead acetate in drinking water (0.05% to 0.45% w/v); dams exposed until weaning; exposure of pups which continued until PND 21, 35, 55, or 85 5 control litters (0%), 10 low-dose litters (0.05%), 8 mid-dose litters (0.15%), 9 high-dose litters (0.45%) (4 male and 4 female pups per litter)	Prenatal Pb exposure that continues until adulthood (85 days old) delays sexual maturation in female pups in a dose-related manner ($p<0.05$); birth weight reduced ($p<0.05$), more pronounced among male pups; decreased growth rates ($p<0.05$) in both sexes accompanied by decrease in plasma concentrations of IGF_1 through puberty ($p<0.05$) and a significant increase in pituitary growth hormone during puberty ($p<0.05$).	PbBs in the pups between the ages of 21 and 85 days were >100 µg/dL and reached up to 388 µg/dL
Ronis et al. (1998c)	Rat/Sprague-Dawley	Lead acetate 0.05, 0.15, or 0.45% in drinking water beginning GD 5 continuing until PND 21, 35, 55, or 85; 5 control litters (0%), 10 low-dose litters (0.05%), 8 mid-dose litters (0.15%), 9 high-dose litters (0.45%) (4 male and 4 female pups per litter)	Dose-responsive decrease in birth weight ($p<0.05$), and crown-to-rump length ($p<0.05$); dose-responsive delay in sexual maturity in male ($p<0.05$) and female ($p<0.05$); neonatal decrease in sex steroids ($p<0.05$); pubertal decrease in testosterone (male) ($p<0.05$), and E_2 (female) ($p<0.05$); decrease estrous cyclicity at high dose ($p<0.05$).	Dams: 0, 48, 88, or 181 µg/dL Pups PND 1: <1, ~40, ~70, or >120 µg/dL Pups PND 21: <1, >50, >160, or ~237 µg/dL Pups PND 35: <1, ~22, >70, or >278 µg/dL Pups PND 55: <1, >68, >137, or ~380 µg/dL Pups PND 85: <1, >43, >122, or >214 µg/dL
Ronis et al. (2001)	Rat/Sprague-Dawley	Lead acetate in drinking water to 825 or 2475 ppm *ad libitum* from GD 4 to GD 55 postpartum; 1 male and female pup/litter (5 litters per group) control group, 1 male and female pup/litter (5 litters per group) 825 ppm Pb-acetate group, 1 male and female pup/litter (5 litters per group) 2475 ppm Pb-acetate group	Dose-dependent decrease of the load of failure in male ($p<0.05$); no difference in plasma levels of vitamin D metabolites; reduced somatic growth ($p<0.05$), longitudinal bone growth ($p<0.05$, and bone strength during the pubertal period ($p<0.05$); sex steroid replacement did not restore skeletal parameters in Pb exposed rats; L-Dopa increased plasma IGF_1 concentrations, rates of bone growth, and bone strength measures in controls while having no effect in Pb exposed groups; DO gap x-ray density and proximal new endostreal bone formation were decreased in the distration gaps of the lead-treated animals ($p < 0.01$); distraction initiated at 0.2 mm 30 to 60 days of age.	PbB at 825 ppm was 67–192 µg/dL PbB at 2475 ppm was 120–388 µg/dL

Table 5-4.3 (cont'd). Selected Studies Showing the Effects of Lead on Mammalian Embryogenesis and Development

Citation	Species/ Strain	Dose/Route/Form/Duration/Group Size	Endpoint/Magnitude of Effect/p-value	Blood Lead Concentration (PbB)
	Rat/Sprague-Dawley	Lead in drinking water at 34 ppm from weaning of mothers through gestation and weaning of offspring until birth; 6 pups control group, 6 pups experimental group	Reduced body weight ($p = 0.04$); parotid function was decreased by nearly 30% ($p = 0.30$); higher mean caries scores than the control pups ($p = 0.005$); pre- and perinatal Pb exposure had significantly increased susceptibility to dental caries ($p = 0.015$).	PbB 48±13 µg/dL

GMP, cyclic guanosine--3',5'-monophosphate; DO, distraction osteogenesis; E_2, estradiol; ERG, electroretinographic; GD, gestational day; IGF_1, insulin-like growth factor 1; LH, luteinizing hormone; NOS, not otherwise specified; PbB, blood Pb concentration; PDE, phosphodiesterase; PND, post-natal day

studies (including those described in the 1986 Pb AQCD and more recent reports) that have demonstrated teratogenic effects of Pb exposure are confounded by maternal toxicity.

5.4.5 Effects on Growth and Endocrine Regulation of Growth

Studies conducted in rodents provide convincing evidence for an association between gestational Pb exposure and reduced birth weight and postnatal growth at doses that produce no clinical toxicity in the mothers (Dearth et al., 2002; Hamilton et al., 1994; Lögdberg et al., 1987; Piasek and Kostial, 1991; Pinon-Lataillade et al., 1995; Ronis et al., 1998a,b,c; Singh et al., 1993b; Watson et al., 1997). In squirrel monkeys, Lögdberg et al. (1987) reported a statistically significant reduction in mean birth weight following oral exposure to Pb-acetate during the latter trimesters of pregnancy (mean maternal PbB 54 μg/dL [39 to 82 μg/dL]). Additional details concerning Lögdberg et al. (1987) are provided in Table AX5-4.3.

In addition, the literature provides convincing support for lead-induced impairment of postnatal growth. Although some early studies (Minnema and Hammond, 1994; Hammond et al., 1993, 1990) ascribed the reduction in postnatal growth to reduced food consumption (suggesting an effect of Pb on the satiety endpoint), more recent studies report impaired growth unrelated to changes in food consumption. Ronis et al. (1996, 1998a,b,c) reported lead-induced reductions in birth weight and postnatal growth that occurred in the absence of a significant alteration in food consumption. Han et al. (2000) found a reduction in the birth length of pups (pup PbB ~16 μg/dL on PND 1) whose mothers had been exposed to Pb up to 1 month before mating (maternal PbB on GD 9, 16, and 21 <40 μg/dL). Berry et al. (2002) reported depressed growth in rats exposed to lead, even though food consumption was higher in the lead-exposed rats.

Ronis et al. (2001) showed that in rats, pre- and postnatal (through PND 55) exposure to Pb reduced somatic longitudinal bone growth and bone strength during the pubertal period (PbB >67 μg/dL). These effects could not be reversed by stimulation of the growth hormone axis by supplemental sex hormone. These results suggest that Pb exposure may impair growth through a mechanism that involves a suppressed pituitary response to hypothalamic stimulation. The mechanism may be related to a reduction in plasma concentrations of IGF_1 following Pb exposure (Dearth et al., 2002; Ronis et al., 1998b). Dearth et al. (2002) exposed F344 rats to Pb by gavage beginning 30 days before mating and continuing until weaning of the pups at 21 days

of age. By PND 30, all three groups had PbB ≤3 μg/dL and all lead-exposed groups exhibited decreased serum levels of IGF_1, LH, and E_2. Since liver IGF_1 mRNA was not affected, it appeared that Pb altered the translation and/or secretion of IGF_1, which in turn decreased LH-releasing hormone at the hypothalamic level. Additional details concerning Dearth et al. (2002) are provided in Table AX5-4.3. An effect on IGF_1 also been demonstrated by Ronis et al. (1998b).

5.4.6 Effects on Other Endocrine Systems during Development

Recent experimental animal studies provide evidence for an interaction between Pb exposure and stress hormones, including glucocorticoids and catecholamines (Cory-Slechta et al., 2004; Yu et al., 1996; Vyskocil et al., 1991; Saxena et al., 1990). Lead has been reported to increase stress hormone levels (Vyskocil et al., 1991).

Cory-Slechta et al. (2004) reported a persistent effect of combined environmental stress (administered as restraint) and maternal Pb exposure (PbB 30 to 40 μg/dL) on corticosteroid levels in adult offspring. Female adult offspring born to these dams exhibited elevated corticosteroid levels only when Pb exposure was combined with environmental stress, whereas their adult male siblings exhibited elevated corticosteroid levels from Pb exposure alone. These data suggest that brief exposures to Pb and stress during development may result in persistent changes in the hypothalamic-pituitary-adrenal axis. Additional details concerning Cory-Slechta et al. (2004) are provided in Table AX5-4.3.

The interplay between Pb and stress hormones is consistent with the findings of Yu et al. (1996) wherein neonatal exposure to Pb (PbB 70 μg/dL) decreased cold-water swimming endurance (a standard test for stress endurance). The enhancement of lead-induced toxicity by stress was also reported by Saxena et al. (1990) in adult male rats. Saxena et al. (1990) reported enhanced testicular injury when rats were exposed to immobilization stress in combination with Pb exposure (PbB >200 μg/dL).

5.4.7 Effects on Other Organ Systems during Development

5.4.7.1 Developmental Effects on Blood and Liver

Recent data provides evidence for lead-induced alterations in developing hematopoietic and hepatic system. The data concerning the effect of Pb exposure on the developing

hematopoietic system are limited. The 1986 Pb AQCD proposed that alterations in blood ALAD activity and erythrocyte protoporphyrin were possible biomarkers for subtle, prenatal effects of Pb on heme synthesis (Hayashi 1983a,b; Jacquet et al., 1977; Prigge and Greve, 1977; Hubermont et al., 1976). A more recent study (Iavicoli et al., 2003) of Pb effects on RBC production, HB concentration, and Hct was not able to clearly establish a dose-response relationship for these endpoints. Although limited by small group size (one litter per dose), subchronic dietary exposure to low levels of Pb (PbB 0.6 to 2 or 2 to 13 µg/dL) revealed that Pb exposure reduced red cell synthesis, hemoglobin concentration, and hematocrit at PbB 2 to 13 µg/dL and increased RBC synthesis, Hb concentration, and Hct at PbB 0.6 to 2 µg/dL. More data are needed to clarify the effect of low-dose Pb exposure on blood endpoints.

Two rodent studies provide limited suggestive evidence that Pb exposure during development produces changes in hepatic enzymes and other biomarkers of hepatic function. Pillai and Gupta (2005) reported that long-term exposure of rats (pre-mating, gestation, and lactation) to moderate levels of Pb-acetate (subcutaneous injections of 0.05 mg/kg-day; PbB not reported) resulted in reduced activities of hepatic steroid (E_2) metabolizing enzymes (17-ß-hydroxy steroid oxidoreductase and UDP glucuronyl transferase) and decreased hepatic CYP450 content. Corpas (2002) reported that exposure to Pb in drinking water exposure during gestation and lactation (pup PbB ~22 µg/dL at PND 12 and PND 21) resulted in alterations in the hepatic systems of neonates (PND 12) and pups (PND 21). The effects manifested as alterations in several biochemical indicators of hepatic toxicity: reductions in Hb, iron, alkaline and acid phosphatase levels, and hepatic glycogen, and elevated blood glucose. These data suggest that Pb may alter hepatic function during development; however, more data are needed to determine whether these effects are persistent.

5.4.7.2 Developmental Effects on Skin

Recent data provides limited evidence of altered soft tissue development resulting from Pb exposure. The literature includes one report of lead-induced abnormalities in skin development. Dey et al. (2001) reported that the pups of mice exposed orally to Pb citrate (5 µg/kg-day) throughout gestation exhibited a variety of skin anomalies, including perforations, cell deformity, and disordered collagen bundles. The PbB levels for mothers and pups were not provided. Although detailed biochemical studies are required to elucidate the mechanism for

structural abnormalities, it appears that covalent binding of Pb ions to the sulfate group of glycosaminoglycans may be involved.

5.4.7.3 Developmental Effects on the Retina

Several studies have found that Pb exposure during early postnatal development impairs retinal development in female Long-Evans hooded rats (Fox et al., 1997, 1991a,b; Fox and Rubenstein, 1989; Fox and Chu, 1988). Of these, two studies are particularly important. Fox et al. (1991a) demonstrated that lactation exposure to Long-Evans hooded rats (PbB 18.8 or 59.4 μg/dL) resulted in long-term, dose-dependent decreases retinal Na/K ATPase activity in the female offspring (only female pups were used). Fox et al. (1997) subsequently demonstrated that lactation exposure to female Long-Evans hooded rats (PbB 19 ± 3 or 59 ± 8 μg/dL) or drinking water exposure to adult females (PbB 56 ± 9 μg/dL) resulted in differential age- and dose-dependent alterations in retinal structure and function following low (PbB <20 μg/dL) and moderate (PbB <60 μg/dL) exposures during lactation or long-term (~60 days) exposure during adulthood. The mode of action for the effects of Pb on retinal development may be related to impaired Na/K ATPase activity (Fox et al., 1991a). The observation of reduced enzyme activity in the retina, but not in the kidney, suggests specificity for the retinal alpha-3 isozyme of Na/K ATPase, rather than the renal alpha-1 isozyme of Na/K ATPase. The authors suggested that this specificity may play a role in the target organ-specific toxicity of Pb (Fox et al., 1991a).

5.4.8 Conclusions

The 1986 Pb AQCD presented unequivocal evidence (derived principally from studies of rodents) for effects of Pb on reproduction and development in laboratory animals. This included evidence for lethal effects in developing organisms exposed to Pb during gestation and in the neonatal period, as well as a variety of sublethal effects on reproduction and development. Sublethal effects included changes in levels or function of reproductive hormones, effects on maturation of reproductive systems, persistent toxic effects on the gonads (both male and female), and adverse effects on the conceptus. More subtle effects on hormone metabolism and reproductive cell structure of developing organisms were also documented.

More recent studies support earlier conclusions, presented in the 1986 Pb AQCD, that Pb can produce temporary and persistent effects on male and female reproductive function and

development and that Pb disrupts endocrine function at multiple points along the hypothalamic-pituitary-gonadal axis.

Effects on Male Reproduction. Studies in experimental animals (presented in the 1986 Pb AQCD and others published subsequent to the 1986 Pb AQCD) provide convincing evidence that Pb acts as an endocrine disruptor in males. The majority of studies support the conclusion that endocrine disruption in males involves Pb acting at multiple sites along the hypothalamic-pituitary-gonadal axis. The adverse effects of Pb on male reproduction include perturbations in sexual development and maturation, changes in fertility, changes in male sex hormone levels, and alterations in gonad tissues and cell structure.

Studies conducted in male experimental animals unequivocally demonstrate that Pb exposure during early development (PbB >30 μg/dL) can delay the onset of puberty and alter reproductive function later in life. Persistent effects of Pb exposure on the male reproductive system may derive from disruption in pulsatile release of sex hormones during early development (Ronis et al., 1998c).

The 1986 Pb AQCD reported evidence that Pb exposure affects male fertility in various animal species, including rabbits (Cole and Bachhuber, 1915), guinea pigs (Weller, 1915), rats (Ivanova-Chemishanska et al., 1980), and mice (Schroeder and Mitchener, 1971). More recent studies, conducted in various animal species, have demonstrated lead-induced alteration of sperm parameters (e.g., count, motility, number of abnormal sperm) (Sokol et al., 1985; Acharya et al., 2003; Adhikari et al., 2000; Foster et al., 1998; Graca et al., 2004; McGivern et al., 1991; Mishra and Acharya, 2004; Sokol and Berman, 1991). These effects, however, have not been observed in all studies; the response may be modified by an adaptive mechanism in the hypothalamic-pituitary-gonadal axis. Lead has also been shown to alter the stability of sperm chromatin in monkeys (PbB 56 μg/dL) in the absence of gross changes in sperm parameters, a finding which may contribute to a reduction in male fertility (Foster et al., 1996a). These results are consistent with observations of reduced in vitro fertilization capacity of sperm collected from rats, rabbits, or mice previously exposed to Pb (Sokol et al., 1994; Foote, 1999; Johansson et al., 1987, respectively). Two modes of action have been proposed for lead-induced alterations in sperm capacity for fertilization: (1) Pb complexation with sulfhydryl groups in sperm, and (2) lead-induced generation of ROS in testes.

Experimental animal studies provide convincing evidence that Pb acts as an endocrine disruptor in males at various points along the hypothalamic-pituitary-gonadal axis. Although there is evidence for a common mode of action, consistent effects on circulating testosterone levels are not always observed in lead-exposed animals. The inconsistency in the reports of circulating testosterone levels complicates the derivation of a dose-response relationship for this endpoint.

The 1986 Pb AQCD reported evidence for histological changes in the testes and prostate in rats in association with relatively high doses of Pb (Chowdhury et al., 1984; Hilderbrand et al., 1973; Golubovich et al., 1968). More recent studies in animals provide additional support for testicular damage (i.e., ultrastructural changes in testes and cytotoxicity in Sertoli cells) following exposure to Pb (Foster et al., 1998; Singh et al., 1993a; Batra et al., 2001; Chowdhury et al., 1986, 1987; Corpas et al., 1995; Graca et al., 2004; Pinon- Lataillade et al., 1993; Saxena et al., 1990). Foster et al. (1998) and Singh et al. (1993a) demonstrated ultrastructural changes in testes of monkeys at PbB 35 to 40 μg/dL. Lead-induced oxygen free radical generation is the plausible mechanism of testicular injury in primates (Foster et al., 1998) and rodents (Chowdhury et al., 1984; Acharya et al., 2003; Adhikari et al., 2001; Batra et al., 2001; Bizarro et al., 2003; Chowdhury et al., 1984; Gorbel et al., 2002; Mishra and Acharya, 2004).

Effects on Female Reproduction. In females, Pb exposure has been consistently shown to disrupt the hypothalamic-pituitary-gonadal axis and to produce reproductive dysfunction. The 1986 Pb AQCD reported that Pb exposure was associated with inhibition of menstruation, ovulation, and follicular growth in monkeys (Vermande-Van Eck and Meigs, 1960) and, in rodents, delayed vaginal opening, decreased frequency of implantation, and reduced rates of pregnancy (Kimmel et al., 1980; Odenbro and Kihlström, 1977). Observations from more recent experimental animal studies support these findings. The effects of Pb on female reproduction may be classified as alterations in female sexual maturation, effects on fertility and menstrual cycle, alterations in levels of female sex hormones, and changes in morphology or histology of female reproductive organs as well as the placenta.

The 1986 Pb AQCD reported that Pb exposure (PbB 20 to 40 μg/dL) in rodents produced delays in sexual maturation. More recent studies in experimental animals (primarily rodent studies) provide convincing evidence that Pb exposure before puberty (prenatal and early postnatal PbB ~40 μg/dL) delays maturation of the female reproductive system (Dearth et al.,

2002, 2004; Iavicoli et al., 2004; McGivern et al., 1991; Ronis et al., 1998a,b,c,). Ronis et al. (1998c) suggested that lead-induced disruption of pulsatile release of sex hormones may result in delayed onset of puberty.

Numerous studies were described in the 1986 Pb AQCD that supported the conclusion that Pb was an endocrine disruptor in females. More recent studies in various mammalian species provide convincing support for endocrine-mediated alterations of the female reproductive system. The nonhuman primate studies provide dose-response information concerning the effects of Pb on female sex hormones and menstrual cycle (Foster et al., 1996b; Foster, 1992; Foster et al., 1992; Franks et al., 1989; Laughlin et al., 1987). Exposures of monkeys to Pb resulting in chronic PbB <20 μg/dL produce few effects on circulating hormone levels and do not alter the menstrual cycle. Higher exposures of monkeys to Pb (PbB >40 μg/dL) alter circulating hormone levels and the menstrual cycle, with more marked changes in these endpoints occurring at higher PbB levels. Several modes of action for lead-induced alterations in female reproduction have been proposed, including changes in hormone synthesis or metabolism (Wiebe and Barr, 1988; Wiebe et al., 1988) and changes in hormone receptor levels (Wiebe et al., 1988; Wide and D'Argy, 1986). In addition, Pb may alter sex hormone release and imprinting during early development (Ronis et al., 1998c; Tchernitchin et al., 1998a,b).

The 1986 Pb AQCD presented convincing evidence from experimental animal studies for lead-induced alterations in female fertility, including interference with implantation and pregnancy. More recent studies have confirmed that Pb exposure disturbs female fertility; however, Pb exposure does not generally produce total sterility. Studies in nonhuman primates and rodents have also demonstrated reductions in litter size, implantation dysfunction, and decreased postnatal survival following Pb exposure of gravid female experimental animals (PbB >30 μg/dL) (Lögdberg et al., 1987; al-Hakkak et al., 1988; Flora and Tandon, 1987; Piasek and Kostial, 1991; Pinon-Lataillade et al., 1995; Ronis et al., 1996; Singh et al., 1993b; Wide, 1985).

Lead-induced changes in morphology or histology in female sex organs and placenta may explain reduced fertility and impaired female reproductive success. Lögdberg et al. (1988) reported a dose-dependent reduction in placental weight and an increase in pathological lesions of the placenta in squirrel monkeys that consuming Pb-acetate in their diet during the last three-fourths or two-thirds of pregnancy (maternal PbB 37 μg/dL). Exposure to Pb in early pregnancy also produces structural changes in the epithelium of the uterus of mice (Nilsson et al., 1991;

Wide and Nilsson, 1979). These changes in uterine tissue may impair successful implantation of the blastocysts. In addition, the histological and morphological effects on the uterus and placenta may explain the reduced birth weight that has been associated with prenatal Pb exposure (possibly due to placental insufficiency).

Developmental Effect. Pre- and postnatal exposure to Pb has been demonstrated to result in fetal mortality and produce a variety of sublethal effects in the offspring. Many of these lead-induced sublethal developmental effects occur at maternal PbB that do not result in clinical (overt) toxicity in the mothers. The few studies that have reported teratogenic effects resulting from Pb exposure are confounded by maternal toxicity.

Studies conducted in rodents and primates provide convincing evidence for an association between Pb exposure and reduced birth weight and postnatal growth at doses that produce no clinical toxicity in the mothers (maternal PbB >40 μg/dL) (Dearth et al., 2002; Lögdberg et al., 1987; Berry et al., 2002; Bogden et al., 1995; Camoratto et al., 1993 Hamilton et al., 1994; Hammond et al., 1989, 1990, 1993; Minnema and Hammmond 1994; Han et al., 2000; Ronis et al., 1996, 1998a,b,c; Piasek and Kostial, 1991; Pinon-Lataillade et al., 1995; Sant'Ana et al., 2001; Singh et al., 1993b; Watson et al., 1997). The available data suggest that the mode of action for lead-induced growth suppression involves a reduction in the plasma concentration of IGF_1.

Recent experimental animal studies provide evidence for an interaction between Pb exposure during development (PbB 30 to 40 μg/dL) and stress hormones, including glucocorticoids and catecholamines (Cory-Slechta et al., 2004; Yu et al., 1996; Vyskocil et al., 1991; Saxena et al., 1990). Lead exposure during early postnatal development (PbB ~20 μg/dL) impairs retinal development in female Long-Evans hooded rats (Fox et al., 1997, 1991a,b; Fox and Rubenstein, 1989; Fox and Chu, 1988).

In addition, recent studies provide limited evidence for lead-induced alterations in developing skin, and hematopoietic and hepatic systems; however, more data are needed to clarify the effect of low-dose Pb exposure on these endpoints.

5.5 CARDIOVASCULAR EFFECTS OF LEAD

5.5.1 Introduction

Numerous large and small epidemiological studies have attempted to examine the link between Pb exposure and development of hypertension (HTN) in the general population and occupationally-exposed individuals. In addition, a number of studies have reported on other cardiovascular effects of Pb in Pb-exposed humans (U.S. Environmental Protection Agency, 1990). While several studies have demonstrated a positive correlation between blood pressure and blood Pb concentration, others have failed to show such association when controlling for confounding factors such as tobacco smoking, exercise, body weight, alcohol consumption, and socioeconomic status. Thus, the studies that have employed blood Pb level as an index of exposure have shown a relatively weak association with blood pressure. In contrast, the majority of the more recent studies employing bone Pb level have found a strong association between long-term Pb exposure and arterial pressure (Chapter 6). Since the residence time of Pb in the blood is relatively short but very long in the bone, the latter observations have provided compelling evidence for the positive relationship between Pb exposure and a subsequent rise in arterial pressure. This section reviews the published studies pertaining to the cardiovascular effects of Pb exposure in experimental animals, isolated vascular tissues, and cultured vascular cells.

5.5.2 Lead Exposure and Arterial Pressure in Experimental Animals

Numerous studies have shown that exposure to low levels of Pb for extended periods results in a delayed onset of arterial HTN that persists long after the cessation of Pb exposure in genetically normal animals (see Tables AX5-5.1 to AX5-5.5). In addition, Pb exposure during gestation has been reported to significantly raise arterial pressure in the third trimester of pregnancy in SD rats given a low calcium diet (Bogden et al., 1995). Taken together, these observations provide irrefutable evidence that extended exposure to low levels of Pb can result in the subsequent onset of HTN in experimental animals.

Many studies have been conducted to explore the mechanisms by which chronic Pb exposure may cause HTN. Most of these studies have examined various blood-pressure regulatory and vasoactive systems in animal models of Pb-induced HTN. In addition, several

studies have investigated the direct effect of Pb on vascular tone or the ability of Pb to modify the response to vasoconstrictor/vasodilator agents in isolated vascular tissues. Finally, a number of studies have explored the effect of Pb on cultured endothelial and vascular smooth muscle cells. An overview of the findings of these studies is provided below:

5.5.2.1 Effect of Lead on Production of Reactive Oxygen Species and Nitric Oxide Metabolism

Reactive oxygen species (ROS), such as, superoxide (O_2^-), hydroxyl radical (OH) and hydrogen peroxide (H_2O_2) are normally produced in the course of metabolism and are safely contained by the natural antioxidant defense system. Excess production and/or diminished containment of ROS can lead to oxidative stress in which uncontained ROS can attack and denature functional/structural molecules and, thereby, promote tissue damage, cytotoxicity, and dysfunction. In fact, oxidative stress has been implicated in the pathogenesis of HTN, atherosclerosis, neurodegenerative disorders, aging, and neoplasm among other afflictions. During the past decade, several studies have demonstrated that Pb exposure causes oxidative stress, particularly in the kidney and cardiovascular tissues, as well as in cultured endothelial and vascular smooth muscle cells (VSMC). The in vivo studies have further shown that Pb-induced oxidative stress is, at least in part, responsible for the associated HTN in experimental animals. Relevant published studies pertaining to this issue are summarized below and listed in Annex Table AX5-5.1.

Khalil-Manesh et al. (1994) were among the first to suggest that oxidative stress may be involved in the pathogenesis of Pb-induced HTN. This assumption was based on the observation that chelation therapy with dimethyl succinic acid (DMSA) rapidly ameliorated HTN and raised plasma cGMP level in rats with Pb-induced HTN. They further demonstrated that DMSA possesses strong antioxidant properties in vitro. Accordingly, they theorized (a) that Pb exposure may increase the generation of ROS, which, in turn, elevate arterial pressure by reacting with and inactivating endothelium-derived-relaxing factor (EDRF), and (b) that by scavenging ROS, DMSA rapidly lowers blood pressure prior to significantly affecting body Pb burden.

In a subsequent study, Gonick et al. (1997) showed a marked increase in renal tissue content of lipid peroxidation product malondialdehyde (MDA) coupled with significant upregulations of endothelial (eNOS) and inducible (iNOS) nitric oxide synthases. Thus, the

study provided evidence for the occurrence of oxidative stress and compensatory upregulation of NOS isotypes in the kidney of animals with Pb-induced HTN.

In another study, Ding et al. (1998) showed that infusion of NOS substrate, L-Arginine, lowers blood pressure to a much greater extent in rats with Pb-induced HTN than that seen in either control animals or DMSA-treated Pb-exposed animals. The data, therefore, provided indirect evidence for the role of depressed NO availability in the pathogenesis of Pb-induced HTN. The study further suggested that oxidative stress may be responsible for diminished NO availability in this model. It should be noted that administrating cell-impermeable native SOD did not lead to a further reduction of blood pressure beyond that seen with L-Arginine alone. As with the previous study (Khalil-Manesh 1994), oral DMSA therapy for 2 weeks significantly lowered blood pressure in the Pb-exposed animals. This was accompanied by a significant reduction of blood Pb concentration. In an attempt to explore whether the observed amelioration of Pb-induced HTN was due to the reduction of Pb burden or alleviation of oxidative stress by DMSA, Vaziri et al. (1997) carried out a study in which rats with Pb-induced HTN were treated with a lazaroid compound, a potent, non-chelating antioxidant. The study revealed marked elevation of blood pressure and oxidative stress (increased lipid peroxidation) and reduced NO availability (depressed urinary $NO_2 + NO_3$ excretion) in the untreated rats with Pb-induced HTN. Antioxidant therapy with the lazaroid compound resulted in a significant alleviation of oxidative stress, improved NO availability, and a marked attenuation of HTN without affecting blood Pb concentration. Thus, the latter study provided convincing evidence for the role of oxidative stress as a major mediator of Pb-induced HTN. The study further demonstrated that Pb-induced HTN is associated with diminished NO availability and that the latter was mediated by oxidative stress. The reduction in NO availability observed in rats with Pb-induced HTN (Pb-acetate, 100 ppm in drinking water for 12 weeks) was recently confirmed by Dursun et al. (2005) in rats treated with daily IP injection of Pb-acetate (8 mg/Kg) for 2 weeks. The authors showed that the rise in arterial pressure was accompanied by a significant reduction of urinary $NO_2 + NO_3$ excretion and a significant fall in renal blood flow (indicating increased renal vascular resistance), mimicking the effect of the NOS inhibitor LNAME.

To further explore the cause for the observed reduction of NO availability, Vaziri et al. (1999a) subsequently studied the expression of eNOS and iNOS in the kidney and cardiovascular tissues of rats with Pb-induced HTN. The study showed that the reduction in NO availability is

paradoxically associated with a significant upregulation of NOS isotypes. Moreover, in vitro incubation experiments revealed no significant change in NOS activity in the presence of lead. Interestingly, antioxidant therapy with pharmacological doses of vitamin E and ascorbic acid reversed the upregulation of NOS isotypes and paradoxically raised NO availability in the subgroup of rats with Pb-induced HTN (Vaziri et al., 1999a). These observations were subsequently confirmed by Vaziri and Ding (2001) who showed marked reduction of NO availability despite significant upregulations of eNOS, nNOS, and iNOS in the aorta, heart, kidney, and brain of rats with Pb-induced HTN and their normalization with the administration of superoxide-scavenger tempol (15 mg/Kg IP/day) for 2 weeks. It is noteworthy that tempol administration had no effect on the measured parameters in the control animals. Taken together, these observations indicated that ROS-mediated NO inactivation and, hence, depressed NO availability, results in a compensatory upregulation of NOS isotypes in animals with Pb-induced HTN. This phenomenon is consistent with other studies from this group, which have demonstrated the presence of a negative-feedback regulation of eNOS by NO (Vaziri and Wang, 1999; Vaziri et al., 2005).

The occurrence of compensatory upregulation of NOS by oxidative stress in Pb-exposed intact animals described above was subsequently replicated by Vaziri and Ding (2001) in cultured human endothelial cells incubated in media containing different concentrations of Pb-acetate (versus control media containing sodium acetate). Once again, co-incubation with tempol prevented this phenomenon. This study confirmed the ability of Pb to affect endothelium independently of its effects on humoral or hemodynamic factors, which are operative in vivo. Taken together, these observations suggest that Pb-induced reduction of biologically-active NO is not due to the reduction of NO-production capacity. Instead, it is linked to oxidative stress. In an attempt to explore this supposition, in a separate study, Vaziri et al. (1999b), tested the hypothesis that avid inactivation and sequestration of NO by ROS may be, in part, responsible for the reduction of NO availability in animals with Pb-induced HTN. To this end, they tested for the presence of immunodetectable nitrotyrosine in kidney, brain, and cardiovascular tissues harvested from untreated and antioxidant-treated (vitamin E + vitamin C) rats with Pb-induced HTN and normal control rats. Nitrotyrosine was used as a marker of NO oxidation by ROS ($NO + O_2^{\bullet} \rightarrow ONOO^-$, $ONOO^- +$ tyrosine $\rightarrow$ nitrotyrosine). The study showed an overabundance of nitrotyrosine in all plasma and tested tissues in the untreated rats with Pb-induced HTN.

Antioxidant therapy reduced nitrotyrosine abundance, attenuated HTN, and simultaneously raised NO availability in the subgroup of rats with Pb-induced HTN but had no effect on the normal control group. These observations provided compelling evidence that Pb-induced HTN causes oxidative stress, which, in turn, promotes functional NO deficiency via ROS-mediated NO inactivation. The latter, in turn, participates in the development and maintenance of HTN and cardiovascular abnormalities. In addition, the formation of the highly cytotoxic reactive nitrogen species, peroxynitrite ($ONOO^-$), from the NO-ROS interaction and the associated nitrosative stress could potentially contribute to the long-term cardiovascular, renal, and neurological consequences of Pb exposure.

In a series of subsequent studies Vaziri et al. (2003) explored the expression of NAD(P)H oxidase (which is a well-recognized source of ROS in, not only, the immune cells but also in renal, cardiovascular, and neuronal tissues) in animals with Pb-induced HTN. In addition, expression of the main antioxidant enzymes, namely Mn and CuZn-superoxide dismutases (SOD), catalase and glutathione peroxidase were investigated. The study revealed significant upregulation the $gp91^{phox}$ subunit of NAD(P)H oxidase in the brain as well as a trend for higher levels in the renal cortex and left ventricle of rats with Pb-induced HTN. This was accompanied by a significant compensatory upregulation of CuZn SOD in the kidney and brain, and of Mn SOD in the heart, of rats with Pb-induced HTN. In contrast, despite the presence of oxidative stress, catalase and glutathione peroxidase activity levels were unchanged. In a more recent study, Farmand et al. (2005), showed a significant increase in CuZn SOD activity with no change in either catalase or glutathione peroxidase activity in the aorta of rats with Pb-induced HTN compared with control animals. Since the latter enzymes are responsible for the reduction of H_2O_2 and lipoperoxides, the lack of an appropriate rise in their tissue levels may contribute to the severity of oxidative stress in Pb-exposed animals.

The contribution of oxidative stress in the pathogenesis of HTN in this model was confirmed by experiments which demonstrated normalization of arterial pressure with the infusion of superoxide-scavenger, tempol, in rats with Pb-induced HTN (but no change was observed in the blood pressure in the control rats) (Vaziri et al., 2003). As noted above, the relative reduction of tissue catalase and glutathione peroxidase, which are responsible for the reduction of H_2O_2 to water and molecular oxygen ($2H2O2 \xrightarrow[GPX]{CAT} 2H2O + O2$), can result in

accumulation of H_2O_2. H_2O_2 serves as a cellular growth signal, as well as a substrate for hydroxyl radical (·OH) generation. The former action can potentially contribute to cardiovascular remodeling, whereas the latter can promote oxidative injury. In a recent study, Ni et al. (2004) demonstrated a transient rise in O_2^- production followed by a sustained rise in H_2O_2 production by human coronary endothelial and vascular smooth muscle cells cultured in media containing Pb-acetate versus the control media containing Na-acetate. This was accompanied by, and primarily due to, upregulation of NAD(P)H oxidase and SOD together with reduced or unchanged catalase and glutathione peroxidase levels. Accordingly, the results of this in vitro study confirmed the findings of the in vivo studies and validated the anticipated accumulation of H_2O_2.

As noted above, H_2O_2 is the substrate for the Fenton and Haber-Weiss reactions, which culminate in formation of the highly cytotoxic·OH ($H_2O_2 + e^- \rightarrow OH + OH^-$). Thus, accumulation of H_2O_2 in animals with Pb-induced HTN can facilitate·OH production and, thereby, promote oxidative stress and tissue injury. This supposition was confirmed in a series of studies by Ding et al. (2001), who showed increased hydroxyl radical production in rats with Pb-induced HTN. Oxidative stress, HTN, and excess hydroxyl radical production were all reversed with IV infusion of the reputed hydroxyl radical scavenger, DMTU, in the Pb-exposed animals. Increased hydroxyl radical production observed in intact animals with Pb-induced HTN was confirmed in lead-treated cultured endothelial cells (Ding et al., 2000). The role of oxidative stress in the pathogenesis of HTN and endothelial dysfunction (depressed NO availability) has been substantiated by a number of other investigators. For instance, Attri et al. (2003), demonstrated that exposure to Pb for up to 3 months resulted in a significant rise in arterial pressure, which was substantially ameliorated by coadministration of the antioxidant vitamin ascorbic acid (20 mg/rat) in Wistar-Kyoto rats. The rise in arterial pressure in lead-treated rats was accompanied by diminished NO availability (low plasma $NO_2 + NO_3$) and biochemical evidence of oxidative stress, i.e., elevations of plasma MDA, a DNA oxidation product (8-hydroxyguanosine), and diminished ferric-reducing antioxidant power, as well as electrophoretic evidence of DNA damage. Amelioration of HTN by antioxidant therapy was accompanied by improved NO availability (plasma $NO_2 + NO_3$), marked attenuation of oxidative stress, and partial reduction of DNA damage in this model. In another study, Malvezzi et al. (2001) showed partial amelioration of HTN in Pb-exposed rats with the administration of either

DMSA or L-arginine and showed a much greater response with the combination thereof. These observations support the role of interaction of ROS and NO in the pathogenesis of Pb-induced HTN in the rat.

As cited above, Pb-induced HTN is associated with and is, at least in part, due to ROS-mediated inactivation and hence, reduced availability of biologically active NO. Many of the biological actions of NO are mediated by cGMP, which is produced from the substrate GTP by the cytosolic enzyme soluble guanylate cyclase (sGC). sGC is expressed in VSMC and several other cell types. The enzyme is activated by NO to produce cGMP, which, in turn, promotes vasorelaxation by lowering cytosolic Ca^{2+} concentrations. In an earlier study, Khalil-Manesh et al. (1993) demonstrated a significant reduction of plasma and urinary cGMP in rats with Pb-induced HTN. These observations prompted a number of studies to evaluate the effect of Pb on sGC expression and cGMP production in vascular tissues obtained from rats with Pb-induced HTN or in normal vascular tissues incubated in Pb-containing media. For instance, Marques et al. (2001) found significant reductions of acetylcholine- and Na-nitroprusside-induced vasorelaxation, despite upregulation of eNOS, in the aorta of rats with Pb-induced HTN. This was associated with marked downregulation of sGC abundance and diminished cGMP production in the aorta. In an attempt to explore the possible role of oxidative stress in Pb-induced downregulation of sGC, they included a group of rats that were co-treated with Pb and the antioxidant vitamin ascorbic acid. Antioxidant therapy ameliorated HTN, restored vasorelaxation response to acetylcholine and Na-nitroprusside, and normalized sGC expression and cGMP production. The authors, therefore, identified diminished sGC as another mechanism by which Pb exposure can promote endothelial dysfunction and HTN. They further showed that Pb-induced downregulation of sGC is mediated by oxidative stress, as evidenced by its prevention with antioxidant therapy. Downregulation of sGC protein abundance in the aorta of Wistar rats with Pb-induced HTN was recently confirmed by Farmand et al. (2005) in the Pb-exposed Sprague-Dawley rats. In another study, Courtois et al. (2003) showed that 24-h incubation of normal rat aorta in the lead-containing media resulted in a concentration-dependent downregulation of sGC (beta subunit), with the maximum effect observed at 1 ppm concentration. This was associated with increased O_2^- production and upregulation of cyclooxygenase-2 (COX-2) expression. Co-incubation with ascorbic acid reduced COX-2 expression and O_2^- production and attenuated, but did not fully prevent, the Pb-induced

downregulation of sGC. Similarly, addition of COX-2 inhibitor Rofecoxib or of protein kinase A inhibitor (H-89) partially mitigated the Pb-induced downregulation of sGC in vitro. However, the COX-2 inhibitor failed to reduce O_2^- production in Pb-exposed vascular tissues. Based on these observations, the authors concluded that Pb exposure downregulates vascular tissue sGC abundance via induction of oxidative stress and upregulation of COX-2.

Oxidative stress and altered NO metabolism can potentially trigger a cascade of events that work in concert to promote HTN and cardiovascular disease in Pb-exposed organisms. Some of these potential links are illustrated in Figure 5-5.1.

5.5.2.2 Protein Kinase C, Inflammation, NFκB Activation and Apoptosis

Protein kinase C (PKC) isoforms belong to a family of serine-threonine kinases, which serve numerous diverse cellular functions. For instance, PKC is involved in regulating vascular contractility, blood flow, permeability, and cell growth. In this regard, the activation of PKC has been shown to cause vascular contraction and Pb exposure has been found to raise PKC activity. For example, Hwang et al. (2002) found increased PKC activity in the erythrocytes of a group of Pb-exposed Korean workers, and Markovac and Goldstein (1988b) showed a significant increase in PKC activity in rat brain micro vessels following exposure to micromolar concentrations of Pb. Also, Watts et al. (1995) demonstrated that Pb-acetate (10^{-10} to 10^{-3} M) caused contraction in an isolated rabbit mesenteric artery preparation. This Pb-induced vasoconstriction was unaffected by denudation of endothelium, while it was significantly potentiated by PKC agonists and attenuated by a PKC inhibitor. Calcium channel blockade with verapamil attenuated, but did not abolish, Pb-induced vasoconstriction. These findings were considered to indicate that activation of PKC is, in part, responsible for Pb-induced vasoconstriction, independently of endothelium or extracellular influx of calcium. Taken together, these observations suggest that the activation of PKC in the vascular smooth muscle cells may, in part, contribute to the pathogenesis of Pb-induced HTN by enhancing vascular contractility. It should be noted, however, that Pb-induced contraction has been shown to be unaffected by a PKC inhibitor in the rat aorta rings (Valencia 2001). Thus, the contribution of PKC activation to the Pb-induced alteration of vascular contractility appears to be both vessel- and species-specific. It is of note, that at high concentrations, Pb can reduce PKC activity in certain cell types, including mouse macrophages and rat brain cortex (reviewed by Watts et al. [1995]).

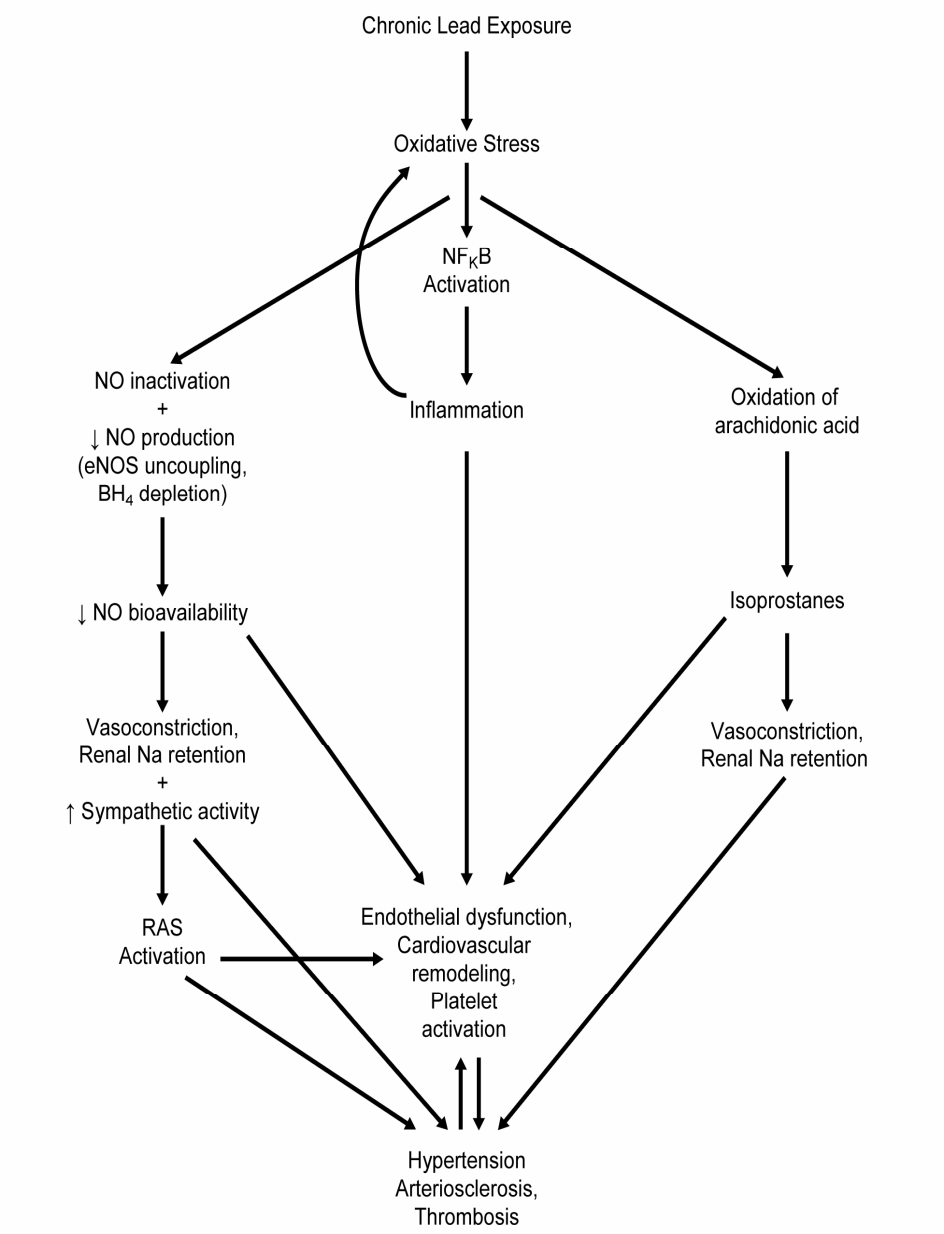

Figure 5-5-1. This illustration depicts some of the potential mechanisms by which oxidative stress may participate in the pathogenesis of Pb-induced HTN and cardiovascular complications. In the presence of oxidative stress, uncontained reactive oxygen species (ROS) inactivate nitric oxide (NO), deplete NO synthase cofactor (tetrahydrobiopterin), uncouple eNOS, promote generation of isoprostanes by oxidizing aracidonic acid, and activate the redox-sensitive transcription factor NFκB. Together, these events can cause vasoconstriction, salt retention, sympathetic system activation, renin-angiotensin system stimulation, platelet adhesion, and, thereby, endothelial dysfunction, hypertension (HTN), inflammation, arteriosclerosis, and thrombosis.

As noted earlier, Pb exposure results in oxidative stress in cultured VSMC and endothelial cells, as well as in intact animals. Oxidative stress can promote the activation of the nuclear transcription factor Kappa B (NFκB) and, thereby, trigger inflammation and apoptosis. In this context, Ramesh et al. (2001) showed that exposure to low Pb levels (50 ppm in drinking water) for 90 days activates NFκB and capsases in the rat brain. It is of note that several studies have revealed the presence of renal tubulointerstitial infiltration of activated T cells, macrophages, and angiotensin II (Ang-II) producing cells in various forms of genetic and acquired HTN in experimental animals. Moreover, the associated tubulointerstitial inflammation has been shown to contribute to the pathogenesis of HTN in these disorders (Rodriguez-Iturbe, 2004). These abnormalities are accompanied by activation of the redox-sensitive NFκB, which can account for the associated inflammation (reviewed by Rodriguez-Iturbe et al. [2004]). The NFκB activation, the accompanying inflammation, and HTN are ameliorated by antioxidant therapy in these models, pointing to the role of oxidative stress in this process. In a recent study, Rodriguez-Iturbe, et al. (2005) observed marked activation of NFκB coupled with tubulointerstitial accumulation of activated T-cells, macrophages, and Ang-II-producing cells, as well as increased apoptotic cells in the kidneys of Pb-exposed rats (100 ppm Pb-acetate in water for 3 months). This was associated with increased nitrotyrosine staining (a marker of NO/ROS interaction) in the kidney tissue. Since tubulointerstitial inflammation plays a crucial role in the pathogenesis of HTN in various other models of HTN, its presence in the Pb-exposed animals may contribute to the associated HTN. Inflammation in Pb-induced HTN is not limited to the kidney. In fact, lymphocyte infiltration is reported in the periaortic tissues in rats with Pb-induced HTN (Carmignani et al 2000). The inflammatory response to Pb exposure in the renal and vascular tissues outlined above parallels the observations reported with immune system in Section 5.9 of this chapter.

5.5.2.3 Effect of Lead Exposure on the Adrenergic System

The adrenergic system plays an important role in regulating arterial pressure, renal and systemic hemodynamics, and cardiac function in health and disease. For this reason, a number of clinical and animal studies have focused on the sympathetic system as a possible mediator of Pb-induced HTN and cardiovascular abnormalities. For instance, in a study of a group of Pb-exposed workers, Chang et al. (1996), found elevated plasma norepinephrine (NE), but normal

plasma dopamine and epinephrine, levels. The constellation of these biochemical abnormalities points to increased sympathetic nervous system activity in Pb-exposed humans. The impact of Pb exposure on the sympathetic nervous system activity has been substantiated in experimental animals. For example, Chang et al. (1997) showed that administration of Pb (Pb-acetate 0.5% in drinking H_2O) for 2 months resulted in significant rises in arterial pressure and plasma NE (but not epinephrine) in Wistar rats. This was coupled with significant reductions of the aorta β adrenergic receptor density and isoproterenol (β agonist)-stimulated cAMP production. In a subsequent study Tsao et al. (2000) reported a significant rise in plasma NE coupled with marked reductions of β receptor density as well as diminished basal and isoproterenol-stimulated cAMP productions in the aorta and heart of Wistar rats with Pb-induced HTN. In contrast to the heart and aorta, β receptor density as well as basal and β agonist-stimulated cAMP production were increased in the kidneys of Pb-exposed animals.

In another study, Carmignani et al. (2000) found significant elevations of blood pressure, plasma catecholamines, and cardiac contractility (dP/dt), together with reduced carotid blood flow in rats with Pb-induced HTN. The effect of Pb on the sympathetic nervous system activity was examined by Lai et al. (2002) who tested the rapid response to intrathecal (IT) injection of $PbCl_2$ in vivo and its addition to the thoracic cord slices in vitro in the rats. They found significant rises in arterial pressure and heart rate with IT injection of Pb-chloride. These effects of Pb were abrogated by the administration of ganglionic blockade using hexomethonium. The in vitro studies revealed a significant rise in excitatory and significant fall in inhibitory post-synaptic potentials with the addition of Pb to the bathing medium and their reversal with saline washout.

In a recent study, Chang et al. (2005) showed a gradual decline in blood, kidney, heart, and aorta Pb contents toward the control values within 7 months following cessation of exposure in rats with Pb-induced HTN. This was coupled with a parallel declines in arterial pressure, plasma NE and renal tissue β receptor density as well as parallel rises in the aorta and heart β receptors densities during the 7-month period following cessation of Pb exposure. However, while HTN and β receptor abnormalities were significantly improved, they were not completely reversed. It should be noted that bone Pb contents were not measured in this study and were most likely elevated despite normalization of blood and soft tissue levels. These findings

provided evidence for the stimulatory effect of Pb on the sympathetic nervous system and for its contribution to the cardiovascular effects of Pb exposure.

5.5.2.4 Effects of Lead on the Renin-Angiotensin-Aldosterone (RAAS) and Kininergic Systems

The available data on the effects of Pb exposure on the RAAS are contradictory. This appears to be primarily due to variability in the dosage and duration of Pb exposure, as well as the age at which exposure is initiated or the animals studied. In addition, when present, nephropathy can potentially affect the RAAS profile of Pb-exposed animals or humans. The majority of animal studies of the effects of Pb on RAAS were conducted and published in the late 1970s and 1980s. In a meta-analysis of the studies published in that period, Vander (1988) found increased plasma renin activity and renal tissue renin content in young rats after several weeks of Pb exposure sufficient to achieve blood Pb concentrations in the range of 30 to 40 µg/dL. Similar results were found in rats exposed to Pb in utero and for 1 month after birth. In contrast, plasma renin activity and renal renin contents were generally unchanged or even reduced in older rats whose Pb exposure had commenced in utero.

In a more recent study, Carmignani et al. (1999) showed a significant increase in plasma angiotensin converting enzyme (ACE) activity in the rats exposed to Pb (60 ppm Pb-acetate in water) for 10 months beginning at an early age (weaning). This was accompanied by a significant increase in plasma kininase II, kininase I, and kallikrein activities. In a subsequent study, Sharifi et al. (2004) examined plasma and tissue ACE activity in young adult rats (weighing 200 g) exposed to Pb (100 ppm Pb-acetate) for 2 to 8 weeks. They found significant rises in plasma, aorta, heart, and kidney ACE activities, peaking at 2 to 4 weeks. This was followed by a decline in plasma and tissue ACE activity to subnormal values by 8 weeks, at which point arterial pressure was markedly elevated. The authors concluded that the elevated ACE activity is involved in the induction of HTN but may not be necessary for maintaining HTN in Pb-exposed animals. Finally, in a recent study, Rodriguez-Iturbe et al. (2005) demonstrated a marked increase in the number of Ang-II positive cells in the kidneys of rats treated with lead-acetate (100 ppm in water) for 3 months. This observation points to heightened intra-renal Ang-II generation in rats with Pb-induced HTN.

Taken together, the data point to activation of the RAAS at some point in the course of Pb-induced HTN. Further studies are needed to fully elucidate the effects of Pb exposure on various other RAAS components.

5.5.3 Effects of Lead Exposure on Vasomodulators

In a study of a group of Pb workers with elevated blood Pb concentration, Cardenas et al (1993) found a significant increase in urinary excretion of the metabolite of vasoconstrictive prostaglandin, thromboxan (TXB_2), and significant reduction of the vasodilatory prostaglandin, 6-keto-PGF1, when compared with the control workers. Subsequently, Hotter et al. (1995) confirmed the elevation of urinary TXB_2 in another group of Pb-exposed workers. Based on these observations, the authors suggested that Pb can alter the balance between vasoconstrictive and vasodilatory prostaglandins in a way which may contribute to HTN and cardiovascular disease. In an attempt to examine such possible effects of Pb exposure in experimental animals, Gonick et al. (1998) measured urinary excretion of the above metabolites in the rat model of Pb-induced HTN. The study showed no significant difference in urinary excretion of the given prostaglandin metabolites between the Pb-exposed and control rats. However, in a recent in vitro study, Dorman and Freeman (2002) demonstrated that Pb promotes the release of arachidonic acid by vascular smooth cells via activation of phospholipase A_2. They further showed that, at low concentrations, Pb augments Ang-II-induced VSMC proliferation, whereas at a high concentration it reduces viability and cell count in unstimulated cells and reduces DNA synthases in Ang-II and Fetal Calf Serum (FCS)-stimulated VSMC. Thus, Pb can increase the release of arachidonic acid (the substrate for prostaglandins) via activation of phospholipase A_2.

Given the limited and contradictory nature of the published data, further in-depth studies are needed to clarify the effects of Pb on regulation of arachidonic acid metabolism and the synthesis of various classes of prostaglandins.

Endothelin

Endothelins (ET) represent a family of potent vasoconstrictive peptides that are produced by endothelium and a number of other cell types. Excess production or increased sensitivity to ET can raise arterial pressure. In an attempt to explore the possible contribution of ET to the pathogenesis of Pb-induced HTN, Khalil-Manesh et al. (1993) studied the effects of exposure to

low and high levels of Pb (100 ppm versus 5000 ppm) in the drinking water for 1 to 12 months in rats. Rats exposed to low (but not high) levels of Pb exhibited HTN and a significant increase in plasma ET-3 concentration. These findings were confirmed by these investigators in a subsequent study of rats with Pb-induced HTN (Khalil-Manesh et al., 1994). Similarly, Gonick et al. (1997) demonstrated a significant elevation of plasma concentration and urinary excretion of ET-3 in rats with Pb-induced HTN. In a recent study, Martin et al. (2005) showed that incubation in the lead-containing media resulted in the downregulation of soluble guanylate cyclase and cGMP production in the isolated artery segment of normal rats. They further found that co-incubation with an ET-A receptor antagonist can partially reverse this effect of lead. These findings suggest that the adverse effect of Pb exposure on cGMP production in the vascular tissue is, in part, mediated by its ability to raise ET activity. It, thus, appears that exposure to low-levels of Pb can raise activity or production of ET, which can, in turn, play a part in the pathogenesis of Pb-induced HTN in the rat. Further studies are required to carefully explore the effects of Pb on various components of the ET system.

Atrial Natriuretic Factor

Atrial natriuretic factor (ANF) is produced and secreted by cardiac myocytes. Plasma concentration of ANF rises with volume expansion and declines with volume contraction. ANF serves as a vasodilator and a natriuretic agent and, as such, plays a role in regulating blood volume, vascular resistance, and, hence, arterial pressure. Giridhar and Isom (1990) measured ANF in rats treated with IP injection of Pb-acetate (0.0 to 1.0 mg/kg/twice weekly for 30 days). The Pb-exposed animals exhibited fluid retention, which was coupled with a paradoxical dose-dependent decline in plasma ANF concentration. Based on these findings, they suggested that Pb may interfere with the hormonal regulation of cardiovascular system, which may, in turn, relate to the cardiovascular toxicity of this metal.

5.5.4 Effects of Lead on Vascular Reactivity

Addition of Pb-acetate to the bathing medium has been shown to elicit a cumulative concentration-dependent vasoconstriction in isolated rabbit mesenteric artery (Watts et al., 1995). This effect was reported to be partly mediated by activation of PKC. In a more recent study, Valencia et al. (2001) found a concentration-dependent vasoconstrictive response to

Pb-acetate (0.1 to 3.1 mM) in Wistar rat thoracic aorta rings. The contractile response was observed in both intact and endothelium-denuded rings. Likewise, Pb-induced vasoconstriction was preserved in calcium-free medium and was unaffected by either α-1 blockade (prazosin), PKC inhibition (Calphostin) or L-type calcium channel blockade (verapamil). However, Pb-induced vasoconstriction was inhibited by lanthanum, which is a general calcium-channel blocker. These observations suggest that Pb can promote an endothelium-independent vasoconstriction by a direct effect on the vascular smooth muscle cells. The data further suggests that the effect of Pb is Ca-independent and may depend on the entry of Pb to the cell via a lanthanum-blockable channel. In contrast to the latter studies, addition of Pb-acetate did not cause vasoconstriction in the rat aorta rings used in a study reported by Shelkovnikov and Gonick (2001). Moreover, Pb-acetate at either high (10^{-4}m) or low (10^{-8}m) concentrations did not modify the response to NE, phorbol ester, or isoproterenol. However, at 10^{-4}M, Pb-acetate augmented the contractile response to submaximal concentrations of calcium. Thus, the rapid action of Pb on vascular reactivity in vitro seems to vary depending on the type of the vessel used, the Pb concentration employed, and the animal species being studied.

A number of studies have endeavored to discern possible differences in vascular reactivity to various agonists between animals with Pb-induced HTN and control animals. For instance, Purdy et al. (1997) found no significant difference in vasoconstrictive response to NE and phenylephrine or vasodilatory response to acetylcholine or nitroprusside in the aorta rings obtained from Sprague-Dawley rats with Pb-induced HTN. In contrast, Marques et al. (2001) showed a significant reduction of vasodilatory response to both acetylcholine and nitroprusside in Wistar rats with Pb-induced HTN. It should be noted that the Wistar rats employed in the latter study had been treated with 5 ppm Pb-acetate in the drinking water for 1 month, whereas those reported by Purdy et al. (1997) had been given a higher dosage (100 ppm) for a longer period (3 months). Therefore, the magnitude and duration of exposure may account for the differences observed between the two reports. Also, the effect of Pb on vascular reactivity may vary from one tissue to the next, as clearly exemplified by studies (Oishi et al., 1996) that showed significant endothelium-dependent vasorelaxation of mesenteric artery response to acetylcholine in the presence of the NOS inhibitor L-NAME in tissues from rats exposed to Pb-acetate for 3 months. These observations suggest that chronic Pb exposure may impair

endothelium-dependent hyperpolarization in the rat mesenteric artery. However, no such effect was noted in the aorta obtained from the same animals.

5.5.5 Lead-Calcium Interactions in Vascular Tissue

Changes in cytosolic Ca^{2+} concentrations are intimately involved in regulating vascular tone and vascular smooth muscle contraction. Consequently, several studies have focused on the interaction of Pb with cellular Ca^{2+} and Ca^{2+}-dependent signaling pathways as a means to gain insight into the pathogenesis of Pb-induced HTN (Piccini et al 1977; Favalli et al 1977; Webb et al 1981; Goldstein 1993; Watts et al 1995). Lead can potentially compete with Ca^{2+} in transport systems (i.e., channels and pumps) involved in physiological movements of ions, particularly Ca^{2+}, into and out of the cell (Simons 1993a,b). Moreover, Pb can alter the intracellular distribution of Ca^{2+} between cytoplasm, endoplasmic reticulum, and mitochondria, which normally regulates cytosolic Ca^{2+} concentration, (Simons 1993a,b). In addition, Pb can serve as a substitute for calcium in Ca^{2+}-dependent signaling pathways by interacting with calmodulin, PKC, and calcium-dependent potassium channels (Haberman, 1983; Richardt, 1986; Chai and Webb, 1988; Simons, 1993a,b; Watts, 1995). Thus, interactions of Pb with cellular Ca^{2+} via these complex mechanisms in the vascular cells may contribute to alterations of vascular resistance and HTN. For example, Piccini et al. (1997) and Favalli et al. (1977) showed that Pb exposure increases calcium content in the tail artery in rats. The authors attributed this phenomenon to a possible Pb-induced inhibition of Ca^{2+} extrusion from the vascular cells. Using rabbit mesenteric artery preparations, Watts et al. (1995), showed that blockade of either PKC or voltage-gated Ca channels by verapamil substantially attenuated Pb-induced vasoconstriction in both intact and endothelium-denuded preparations. Based on these observations, the authors suggested that Pb promotes a vasoconstrictive response in rabbit mesenteric artery via a Ca^{2+}-dependent activation of PKC. In contrast, Valencia et al. (2001) using rat aorta rings reported a vasoconstrictive response to Pb-acetate in rat aorta rings bathed in either Ca^{2+}-free or Ca^{2+}-containing media and in the presence or absence of the L-type calcium-channel blocker verapamil or of the PKC inhibitor calphostin. Moreover, depletion of intracellular Ca^{2+} stores by preincubation of rings in EGTA, while diminishing the intensity, did not abrogate Pb-induced vasoconstriction in this system. In contrast, Pb-induced vasoconstriction was prevented by lanthanum (a general blocker of calcium channels) in both Ca^{2+}-containing and Ca^{2+}-free media.

Based on these observations, the authors concluded that Pb can elicit a PKC-independent contractile response in the rat aorta by entering VSMC via a non-voltage-gated Ca^{2+} channel and mimicking the action of Ca^{2+}. It, thus, appears that Pb exerts its effect by mechanisms that are species- and vessel-specific.

5.5.6 Cardiotoxicity and Atherogenesis

Acute Pb exposure has been reported to affect cardiac function, and chronic exposure has been linked to atherosclerosis and increased cardiovascular mortality by some, but not by all investigators, in humans (See Chapter 6). In an attempt to assess the cardiotoxicity of lead, Prentice and Kopp (1985) carried out the in vitro perfusion of isolated rat heart preparations with a perfusate containing 0.3 and 30 μM Pb-acetate for up to 60 min. At 30 μM concentration, Pb prolonged the AV node and His bundle conduction times, reduced coronary blood flow and heart rate, and altered cardiac energy metabolism. Milder, and statistically insignificant, changes were also observed at 0.3 μM Pb concentration in this model. These observations illustrate the direct cardiotoxicity of Pb independently of its systemic and neuroendocrine actions in acute intoxication. In an attempt to determine whether chronic exposure to Pb or cadmium can cause atherosclerosis, Revis et al. (1981), studied male white pigeons that were exposed to Pb (0.8 ppm in drinking water) for extended periods. Long-term low-level Pb exposure in this model resulted in a significant rise in arterial pressure and a near doubling of the number of atheromatous plaques in the aorta. These observations demonstrate the proatherogenic effects of chronic exposure to low levels of Pb in pigeons.

5.5.7 Effects of Lead on Endothelial Cells

Endothelium is an important constituent of the blood vessel wall and regulates macromolecular permeability, vascular smooth muscle tone, tissue perfusion, and blood fluidity. Endothelial damage or dysfunction results in atherosclerosis, thrombosis, and tissue injury. Chronic Pb exposure has been shown to promote atherosclerosis in experimental animals (Revis et al., 1981). Given the central role of endothelial injury/dysfunction in the pathogenesis of atherosclerosis, numerous studies have explored the effect of Pb on cultured endothelial cells. These studies have searched for evidence of Pb-mediated endothelial cell injury and the effects of Pb on endothelial cell proliferation, tube formation (angiogenesis), monolayer wound repair,

and production of heparansulfate proteoglycans, plasminogen activator (tPA), and plasminogen activator inhibitor-1 (PAI-1).

Using cultured bovine aorta endothelial cells, Kaji et al. (1995a) showed that incubation with Pb-nitrate at concentrations equal to or below 50 μM for 24 h, results in mild de-endothelialization of endothelial monolayers in vitro. They further showed that adding Pb at 10 μM concentration markedly increased cadmium-induced endothelial injury.

Proliferation of endothelial cells is a critical step for the repair of injured endothelium. Failure of the repair process can result in thrombosis, VSM cell migration and proliferation, and atherosclerosis. In this regard, Pb (Pb-nitrate 0.5 to 5 μM) has been shown to significantly reduce DNA synthesis and cell proliferation in growing cultured bovine aorta endothelial cells (Kaji, 1995a). Similarly, the proliferative response to βFGF and αFGF is significantly attenuated by Pb in this system (Kaji, 1995b). The reported inhibition of endothelial cell proliferation by Pb can potentially diminish the repair process in response to endothelial injury. This supposition has been confirmed by Fujiwara et al. (1998) who showed that at 5 to 10 μM concentrations, Pb markedly inhibited the repair of the wounded endothelial monolayer in vitro. Moreover, Pb severely mitigated the zinc-stimulated endothelial cell proliferation and repopulation of the denuded sections in this system.

Endothelial cell proliferation is the primary step in angiogenesis, a phenomenon that is essential for numerous physiological functions such as growth, development, wound repair, and menstrual cycle as well as certain pathological events including diabetic retinopathy and tumor growth. In view of the demonstrated inhibition of endothelial cell growth by lead, it has been postulated that Pb may impair angiogenesis. This assumption has been confirmed by a number of studies testing the effect of Pb by angiogenesis assay (tube formation) in endothelial cells cultured on matrigel (a laminin-rich basement membrane product) matrix in vitro. For instance, Ueda et al. (1997) and Kishimoto et al. (1995) have shown that Pb-acetate (1 to 100 μM) results in a concentration- and time-dependent inhibition of tube formation by human umbilical vein endothelial cells cultured on a matrigel matrix.

Endothelial cell migration and proliferation are critical for angiogenesis and repair of the damaged endothelium. βFGF is a powerful mitogen for endothelial cells as well as several other cell types. Endothelial cells synthesize βFGF, which is released following injury or spontaneous death of endothelial cells and acts in an autocrine fashion to facilitate the repair process by

promoting endothelial cell migration and proliferation. Binding of βFGF to its receptor on the endothelial cell is facilitated by heparan sulfate proteoglycans (HSPGS) that are normally produced and released by the endothelial cells for attachment to the cell surface as well as incorporation in the extracellular matrix. As noted above, Pb significantly attenuates βFGF and αFGF-mediated DNA synthesis and proliferation in cultured endothelial cells (Kaji et al., 1995b). In this regard, Pb has been shown to reduce βFGF binding to the cell surface HSPGs without changing the biosynthesis or intracellular abundance of βFGF in cultured bovine endothelial cells (Fujiwara and Kaji, 1999a). Moreover, Pb has been shown to significantly reduce the synthesis of glycosamino-glycans (GAG, measured by sulfate incorporation into heparan sulfate) in the growing endothelial cells.

The above observations suggest that Pb-induced reduction of βFGF-mediated proliferative response in cultured endothelial cells is largely due to impaired production of HSPGs. This supposition is further supported by observations that DNA synthesis can be restored by adding heparin in lead-treated growing endothelial cells (Fujiwara et al., 1995). The reduction in the production of GAGs by Pb in the growing endothelial cells (Fujiwara et al., 1995) is also seen in confluent (quiescent) cells. For instance, Kaji et al. (1991) demonstrated a marked reduction of GAG production following incubation with 10 μM Pb nitrate in confluent endothelial cells in vitro. The Pb-induced reduction of heparan sulfate production was more severe than that of the other GAGs. Moreover, the reduction in the cell surface-associated GAGs was more severe than that of the newly synthesized GAG found in the incubation media. GAGs combine with a series of specific core proteins to form anionic macromolecular complexes known as proteoglycans, which are widely distributed in the extracellular matrix of the mammalian tissues. Endothelial cells produce two types of HSPGs, i.e., the high-molecular weight and low-molecular weight classes. Perlecan is a high-molecular weight heparan-sulfate proteoglycan which is a component of the basement membrane. Syndecan, glypican, ryudocan, and fibroglygan are among the low-molecular weight subclass and are primarily associated with the cell surface. Proteoglycans play an important role in regulating vascular function and structure. For instance, by providing a negative electrostatic charge, these molecules constitute a major barrier against extravasations of negatively-charged plasma proteins. In addition, by interacting with antihthrombin-III and tPA, these molecules serve as important endogenous anticoagulants. Moreover, perlecans facilitate βFGF binding to its receptor on endothelial cells and, thus, contributes to the endothelial growth

and repair processes. In contrast, these molecules tend to inhibit migration and growth of vascular smooth muscle cells and, thereby, help to prevent athero- and arteriosclerosis. Another important function of HSPGs is their role in stabilizing and anchoring lipoprotein lipase and VLDL receptors on the endothelial surface. Consequently, they play an important indirect part in the clearance of VLDL and chylomicrons from the circulation, a process which has major implications for energy metabolism and cardiovascular protection.

In a study of cultured bovine endothelial cells, Kaji et al. (1997) found that Pb-chloride, at 10 μM concentration, markedly lowers incorporation of precursors (glycosamine and sulfate) into HSPG in confluent bovine aorta endothelial cells. The effect of Pb was more severe on low-molecular than high-molecular weight HSPGs. However, Pb did not change the length of heparan sulfate chains. It is of note that Pb slightly increased the abundance of the HSPG core proteins. This observation excluded a reduction in core protein synthesis as a cause of diminished HSPGs in the lead-treated confluent endothelial cells. In a subsequent study, Fujiwara and Kaji (1999) investigated the effect of Pb-nitrate on production of high- and low-molecular weight subclasses of HSPGs in growing bovine aorta endothelial cells. In contrast to the quiescent cells, lead-treated growing cells exhibited a marked reduction in the high-molecular weight with no change in production of low molecular weight (~50KD) HSPGs. They further showed a significant reduction of the core protein of perlecan, which is a high-molecular weight (400 KD) HSPG. Thus, Pb appears to affect productions of subclasses of HSPGs differently depending on the cells' growth cycle. Accordingly, in the growing endothelial cells (a condition which simulates the response to injury), Pb downregulates perlecan, which is involved in βFGF-mediated migration and proliferation of endothelial cells and inhibition of migration and proliferation of VSMC. This phenomenon may adversely affect endothelial repair and promote athero- and arteriosclerosis. On the other hand, Pb-induced reduction of the cell surface-associated low-molecular weight HSPGs (which are predominantly involved with lipolytic, anticoagulant, and other functions of confluent endothelial cells (simulating intact endothelium) can contribute to hyperlipidemia and thromboembolism, among other disorders.

One of the major properties of normal endothelium is its ability to prevent coagulation. Several factors contribute to the thromboresistance of the endothelial lining. These include the surface coating of HSPG (which confers heparin-like properties), nitric oxide (which inhibits platelet adhesion and activation), and tPA (which promotes thrombolysis), thrombomodulin, and

prostacycline. As noted earlier, Pb exposure reduces HSPG-production (Kaji et al., 1995b, 1997) and diminishes nitric oxide availability via ROS-mediated NO inactivation (Vaziri 1999). In addition, Kaji et al. (1992) showed that incubation of confluent human umbilical vein endothelial cells with Pb nitrate, at 0.01 to 1.0 μM concentrations, significantly reduced basal and thrombin-stimulated tPA release. It thus, appears that Pb exposure may confer a thrombophilic diathesis.

5.5.8 Effects of Lead on Vascular Smooth Muscle Cells

Lead has been shown to stimulate proliferation of bovine aorta VSMCs in a concentration-dependent manner (Fujiwara et al., 1995). Moreover, the combination of Pb and βFGF results in an additive effect on VSMC proliferation. As with bovine aorta VSMCs, cultured rat aorta VSMCs exhibit hyperplasia in response to a low concentration of (100 μg/L) of Pb-citrate (Carsia, 1995). The reported hyperplasia is accompanied by phenotypical transformation of cells from the spindle or ribbon shape to cobblestone shape, simulating the neointimal cell morphology. This was accompanied by a significant reduction in Ang-II receptor but no change in α, β, or ANP receptor densities. It is of note that, in contrast to the low concentration, a high concentration (500 μM/L) of Pb resulted in growth arrest in this system. Thus, the effect of low concentration of Pb on VSMC proliferation is opposite of its action on the endothelial cells.

Under normal conditions, intact endothelial lining shields the cells residing in the subendothelial tissue, i.e., fibroblasts and VSMCs, from coming into contact with the circulating blood. However, this barrier is lost when the endothelium is injured, an event which can lead to platelet adhesion and fibrin thrombosis formation. Propagation of fibrin thrombus is limited by activation of the fibrinolytic system, which, in turn, depends on the balance between tPA and plasminogen activator inhibitor-1 (PAI-1). In addition to endothelial cells, VSMCs and fibroblasts express tPA and PAI-1. Using cultured human aorta VSMCs and fetal lung fibroblasts, Yamamoto et al. (1997) investigated the effect of Pb chloride on the release of tPA and PAI-1 in vitro. The authors found that Pb causes a significant inhibition of tPA release and a significant increase in PAI-1 release in cultured fibroblasts in a dose-dependent manner. The lead-treated VSMC exhibited a significant dose-dependent decline in tPA release and to a lesser extent of PAI-1 release. Taken together, exposure to Pb appears to evoke a negative effect on fibrinolytic process by the cellular constituents of the subendothelial tissue.

5.5.9 Summary/Conclusion

In vivo and in vitro studies published during the past 15 years have considerably expanded our knowledge of the effects of Pb exposure on the cardiovascular system. However, many questions remain unanswered and await further investigation.

A number of in vivo and in vitro studies conducted during the review period have provided compelling evidence for the role of oxidative stress in the pathogenesis of Pb-induced HTN. Moreover, the effect of oxidative stress on blood pressure has been shown to be, in part, mediated by avid inactivation of NO and downregulation of sGC. In addition, a limited number of in vitro studies have provided indirect evidence that, via activations of PKC and NFκB, Pb may raise vascular tone and promote inflammation.

The adrenergic system plays a major role in regulating cardiovascular function and structure and, as such, has been the focus of several studies during the review period. Based on these studies, chronic low level lead exposure appears to increase central sympathetic activity, reduce cardiac and vascular and raise kidney β adrenergic receptor density. These events can, in turn, increase peripheral vascular resistance and renal renin release/production and, thereby, arterial pressure. Since sympathetic outflow is inhibited by NO, inactivation of NO by oxidative stress may be, in part, responsible for the increased sympathetic activity in Pb-exposed animals.

The renin-angiotensin-aldosterone system (RAAS) plays an important role in regulating blood pressure and cardiovascular function and structure. The available data published during the review period suggest that Pb exposure can raise plasma ACE and kininase activities at different points in the course of Pb-induced HTN in experimental animals. This can, in turn, contribute to the genesis and/or maintenance of HTN. Since renin release (which is responsible for production of ACE substrate, i.e., Ang-1) is, in part, driven by β adrenergic activation, upregulation of renal β adrenergic activity may, in part, account for increased RAAS activity in the Pb-exposed animals.

The balance in production of vasodilator and vasoconstrictor prostaglandins plays an important role in regulation of blood pressure and cardiovascular function. Studies of the Pb exposed humans have revealed an imbalance in production of prostaglandins favoring a rise in arterial pressure. However, the animal and in vitro studies published during the review period have been limited and inconsistent. Further studies are needed to address this issue.

Based on the available studies, Pb exposure appears to increase endothelin production in experimental animals. This phenomenon can, in part, contribute to the rise in blood pressure in the Pb-exposed animals. A number of studies have explored the effect of Pb on vascular tone as well as vascular response to vasoconstrictor and vasodilator agents. For instance, Pb has been shown to cause vasoconstriction and to attenuate acetylcholine- and NO-mediated vasodilatation in some, but not all vascular tissues and in some, but not all, studies. These effects have been variably attributed to lead-mediated activation of PKC and Ca^{2+}-mimetic action of Pb, among other possibilities.

Finally, a number of studies have explored the effects of endothelial and vascular smooth muscle cells to explore the possible atherogenic effect of Pb exposure. In this context, Pb has been found to inhibit proliferation of the growing (non-confluent) endothelial cells (mimicking in vivo response to injury), impair tube formation (angiogenesis), and the repair of wounded endothelial monolayer in vitro. Likewise, Pb exposure was shown to reduce production of HSPGs and tPA by confluent endothelial monolayers, events that may favor thrombosis and hyperlipidemia. Lead exposure has been also shown to promote vascular smooth muscle cell and fibroblast proliferation and phenotypic transformation in ways that seem to favor arteriosclerosis and vascular remodeling.

Among many questions awaiting clarification, a few are of particular interest. For instance, it is not clear as to why low, but not high, levels of Pb exposure cause HTN in experimental animals. Similarly, it is uncertain as to why HTN occurs long after the onset of Pb exposure in the intact animals, whereas the effects on cultured cells and isolated tissues are manifested within short periods of time.

5.6 GENOTOXIC AND CARCINOGENIC EFFECTS OF LEAD

5.6.1 Introduction

The 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986) and its 1990 Supplement (U.S. Environmental Protection Agency, 1990) concluded that, at relatively high concentrations, Pb may be carcinogenic to laboratory animals, particularly the rat. Cell culture studies were considered to be supportive of these observations, but also indicated that Pb was not

particularly potent. Human data were considered to be of concern, but not definitive, and given the animal data, the prudent choice was to consider Pb to be a possible human carcinogen.

This section reviews reports of Pb-induced carcinogenesis and DNA damage published since 1986. More than 200 publications were read and considered and those that reported any effect related to carcinogenesis or genotoxicity that was attributable to Pb are presented below.

This report follows the same format as the previous one (1986) and the explanations for the relative importance of the various types of studies (e.g. epidemiology, animal and cell culture) can be found in the original report and are not repeated here. Carcinogenesis studies are presented first, followed by genotoxicity studies. Each of these sections is further subdivided into human studies (considering adults and then children), animal studies, and then cell culture studies (considering human, mammalian, and then nonmammalian). When appropriate, these sections are followed by a section describing acellular (cell-free) model studies.

There are some differences with this new report. For one, each section is more distinctly broken out. The epidemiology has been reviewed in more detail in Chapter 6 (Section 6.7) in this document and, so, only a brief summary is presented here. Because of more recent concerns about effects on childhood development, this issue was specifically considered in a separate section. Following advances in hypotheses and technology, much more specific sections about the possible epigenetic effects of Pb have also been added.

5.6.2 Carcinogenesis Studies

5.6.2.1 Human Studies

The human carcinogenesis studies are only briefly reviewed in this section; for a more detailed review, see Chapter 6 (Section 6.7) in this document.

Adults

The assessment of the carcinogenicity of Pb through human epidemiological studies remains ambiguous. Several reports state that occupational exposure to Pb increases the risk of lung, kidney, brain, stomach, and liver cancer (Fu and Boffetta, 1995; Kauppinen et al., 1992; Gerhardsson et al., 1995a; Ades and Kazantzis, 1988; Wicklund et al., 1988; Steenland et al., 1992; Englyst et al., 2001; Gerhardsson et al., 1986; Antitila et al., 1995, 1996; Cocco et al., 1998; Shukla et al., 1998). However, a full interpretation of the data in these studies is

complicated by the fact that the study participants also incurred coexposure to other known carcinogens, such as arsenic, cadmium, and hexavalent chromium. Thus, it is difficult to determine if the excess cancers observed were due to exposure to Pb, one of these other carcinogens, or some combination of the various chemicals. In addition, other reports indicate that occupational or environmental exposure to Pb did not alter cancer risk (Cocco et al., 1996; Fanning, 1988; Jemai et al., 2002). Consequently a definitive assessment of the carcinogenicity of Pb from human studies cannot be made at this time.

Children

There have been no recent studies of Pb-induced cancers in children. This lack of data is not unexpected and is largely because Pb has not been considered a likely cause of childhood cancers. There have, however, been studies of cancers in children resulting from paternal exposure. Here again, the same confounding problems encountered are as seen in the adult population studies, and it is difficult to draw any definitive conclusions. For example, two studies reported elevated childhood tumors (Wilm's tumor and acute nonlymphocytic leukemia) in children whose fathers worked in Pb-related industries, such as welding, painting, and auto repair (Buckley et al, 1989; Olshan et al., 1990). However, workers in these occupations also experienced coexposure to arsenic, cadmium, and hexavalent chromium, and so the cancers observed cannot be solely linked to Pb exposure. In addition, a report from the printing industry in Norway found no link between paternal exposure and childhood cancers and, perhaps, even found a possible reduction in the incidence of childhood cancers with paternal Pb exposure (Kristensen and Andersen, 1992).

The possible interaction of paternal occupation and childhood cancer is an important area of concern. However, a definitive assessment of paternal exposure to Pb cannot be made at this time and more research is needed.

5.6.2.2 Laboratory Animal Studies

Lead is a well-established animal carcinogen, as noted in the 1986 Lead AQCD. Consequently, limited tumorigenesis studies have been conducted in animal models and the focus has been more on the mechanism of neoplasia (e.g., the roles of calcium and

metallothionein) and possible immunomodulatory effects of Pb in the promotion of cancer. These studies are summarized in Table AX5-6.1.

All of the studies exposed animals to Pb-acetate except one, which focused on Pb-chromate. One study investigated the carcinogenicity of a series of chromate compounds, i.e., Pb-chromate and several Pb-chromate-based compounds were included as part of the group of chromate compounds. The Pb-chromate was administered by implantation into the lung after being embedded within a cholesterol pellet. The authors indicated that in this design, Pb-chromate was not carcinogenic, but that 4 of the Pb chromate compounds did induce a very rare tumor in the mice. Thus, there is some ambiguity about the carcinogenicity of Pb-chromate in the study, as the statistics calculated an expected tumor level based on any tumor and were not based on the occurrence of this very rare (for rats) tumor. It is likely that had the expected value been adjusted for the rare tumor, a conclusion would have been reached that either Pb-chromate was tumorigenic or that the study lacked the power to make any determination. The previous EPA report had concluded that Pb-chromate is tumorigenic. Thus, it is difficult to draw a firm conclusion from this study.

The remaining five studies focused on Pb-acetate (Schrauzer, 1987; Blakley, 1987; Teraki and Uchiumi, 1990; Bogden et al., 1991; Waalkes et al., 2004). In most studies, this compound was administered in drinking water at concentrations from 0.5 to 4000 ppm, but one study considered effects from a subcutaneous (SC) injection both in mice and in rats. Consistent with the findings in the 1986 Pb AQCD, Pb not only induced renal tumors, but also induced other tumors, although the possible effect on mammary tumors is difficult to interpret, as important study details were omitted, as discussed below. In a surprising development, during one lifetime exposure study, Pb suppressed liver tumors (Waalkes et al., 2004).

The key study in this group of studies was a lifetime exposure study that investigated mice exposed to drinking water concentrations of 1,000 to 4,000 ppm Pb and also considered the role of metallothionein. In wild-type mice, Pb-acetate induced a low frequency of renal tumors, but hyperplasia was common and exhibited overexpression of cyclin D1. Lead inclusion bodies were also common. Lead also suppressed liver tumors in this study.

By contrast, in metallothionein-deficient mice, Pb-acetate induced a high frequency of kidney tumors and severe inflammation. Both the tumors and the regions of inflammation exhibited cyclin D1 overexpression. Lead also suppressed liver tumors in these animals. In

contrast to the wild-type mice, Pb inclusion bodies were not seen in these animals. Thus, the data convincingly indicate that metallothionein binds Pb as part of an inclusion body and prevents the formation of tumors.

Another study focused on the ability of Pb to induce tumors in rats after SC injection of Pb-acetate (Teraki and Uchiumi, 1990). Tumors formed at the site of injection, and Pb accumulated in the tumors, indicating that Pb is tumorigenic. However, full interpretation of the data is complicated by the absence of data on control animals and the fact that only a single dose was considered.

Three studies investigated compounds that might reduce or prevent Pb-induced cancers, specifically selenium and calcium compounds (Schrauzer, 1987; Bogden et al., 1991). The first study used a rather complex approach to study the possibly protective effects of selenium (Shrauzer, 1987). In this study, mice were infected with the murine mammary tumor virus, because they are known to develop mammary adenocarcinomas when maintained on a low-selenium diet. The data indicated that Pb can induce tumors in these mice even when they are maintained on a high-selenium diet. However, the data are difficult to interpret and the impact of the study is uncertain, as the methods are incomplete, the data on control animals are not provided, and the experimental results are stated but not presented in tables or figures.

The second study investigated the effect of calcium (Bogden et al., 1991). The main focus of this study appeared to be blood pressure, but tumorigenesis was also considered. It might be anticipated that calcium might reduce Pb tumorigenesis by competing for its binding sites or blocking its uptake. However, in this study, calcium did not affect Pb levels in tissue and actually exacerbated Pb-induced carcinogenesis. The full impact of this study is also difficult to assess, as the calcium-treated animals incurred profound nephrocalcinosis.

The remaining study considered Pb-induced immunosuppression as a possible factor contributing to the tumorigenesis induced by other agents, including viruses or chemicals (Blakley, 1987). The results indicated that Pb may suppress humoral immunity but not cellular immunity. However, this is the only study of its kind and the results need to be repeated in other settings. In addition, it is difficult to determine if these data are specific to the agents used (e.g., murine lymphocytic leukemia virus) or if they represent a class of agents (e.g., viruses in general).

Overall, the above studies confirm that Pb is an animal carcinogen and extends our understanding of mechanisms involved to include a role for metallothionein. Specifically, the recent data show that metallothionein may participate in Pb inclusion bodies and, thus, serves to prevent or reduce Pb-induced tumorigenesis. Much more work is needed to determine the potential exacerbating or ameliorating roles of calcium and selenium and to determine what role Pb-induced immunomodulation may play in the promotion of tumors.

5.6.2.3 Cell Culture Studies

Carcinogenesis is measured in cell culture systems through studies of neoplastic transformation, where morphologically transformed cells are injected into athymic mice to see if the cells can form a tumor in the host animal. Morphological transformation refers to cells that incur a change in morphology, such as formation of a focus (or foci) of cell growth. In addition, for faster study results and as a screening tool, the ability of cells to grow in agar without a surface to attach to (anchorage independence) is often used as a short-term substitute measure for transformation.

Human Cell Cultures

Since the 1986 Pb AQCD, only four studies have used human cell culture systems to study the carcinogenesis of Pb compounds. One found that Pb-acetate induced anchorage independence in primary human foreskin fibroblasts (HFF) (Hwua and Yang, 1998). The full impact of these data is uncertain, as previous studies of known metal carcinogens in primary HFF found that these carcinogens induced anchorage independence, but those anchorage-independent cells ultimately senesced. These studies are summarized in Table AX5-6.2. Further study is needed to confirm that Pb can induce anchorage independence and to see if these cells can progress to full neoplastic transformation.

In an effort to explore the importance of oxidative metabolism in inducing anchorage independence, Hwua and Yang (1998) also co-treated some cells with 3-aminotriazole, a known catalase inhibitor. This co-treatment had no effect on Pb-acetate-induced anchorage independence, suggesting that catalase was not involved in this effect. It would be premature to conclude that oxidative metabolism is not involved in anchorage independence, as these are the

only data available and are limited to catalase only. More data are needed to elucidate whether oxidative metabolism is involved in this lead effect.

The remaining three studies focused on Pb-chromate (Beiderman and Landolph, 1987, 1990; Sidhu et al., 1991). Two used similar HFF cells and found that Pb-chromate induced anchorage independence (Beiderman and Landolph, 1987, 1990). However, these anchorage-independent cells ultimately underwent senescence, suggesting that anchorage independence may not be a suitable short-term marker for neoplastic transformation in primary HFF. It should be noted that these studies were focused on the chromate component of this compound and the potential contribution of Pb was not investigated or discussed. By contrast, Sidhu et al. (1991) found that Pb-chromate did not induce anchorage independence in a human osteosarcoma cell line, while it did induce full neoplastic transformation of these cells and the transformed cells did grow in agar. It should be noted that this study was also focused on the chromate component of this compound and that the potential contribution of Pb was not investigated or discussed.

The 1986 Pb AQCD did not include any studies of transformation in human cells. Given that other chromate compounds have been shown to induce anchorage independence, it seems quite possible that the data from Pb-chromate exposures may represent effects from chromate and not from Pb. Thus, the data currently seem to indicate that Pb can induce anchorage independence in human cells, but its ability to induce neoplastic transformation of human cells is uncertain. Further study of different Pb compounds and the full assessment of their neoplastic potential (i.e., including studies of the ability of treated cells to form tumors in experimental animal models) are needed before definitive conclusions can be drawn.

Animal Cell Cultures

The 1986 Pb AQCD presented several studies demonstrating that Pb compounds could induce anchorage independence and morphological and neoplastic transformation in rodent cell culture systems. Since that report, six studies have further considered the ability of Pb compounds to induce these effects. Three focused on Pb-chromate and three on Pb compounds without the confounding factor of chromate; and these studies are summarized in Table AX5-6.3.

Four studies considered Pb-acetate, Pb-chloride, or Pb-nitrate in Syrian hamster embryo and C3H10T1/2 mouse embryo cells (Zelikoff et al., 1988; Patierno et al., 1988; Patierno and Landolph, 1989; Elias et al., 1991). Three found that Pb compounds did not induce

transformation (Patierno et al., 1988; Patierno and Landolph, 1989; Elias et al., 1991); but the third study (Zelikoff et al., 1988) indicated that Pb was weakly positive, though no statistics were performed to validate this conclusion. Zelikoff et al. (1988) indicated that the observations were repeated several times, but only showed data from one experimental run. It is unclear why the studies were not averaged together, as multiple repeats would likely have provided the power to detect whether the observed weak increase was significant.

Five studies considered Pb-chromate, which induced neoplastic and morphological transformation of Syrian hamster and mouse C3H10T1/2 embryo cells, as well as enhancing viral transformation (Patierno et al., 1988; Patierno and Landolph, 1989; Schectman et al., 1986; Elias et al., 1989, 1991). The focus on Pb-chromate was based largely on concern about chromate; but these studies found that Pb-chromate was more potent than other chromate compounds, suggesting that Pb may enhance or contribute to the carcinogenicity. Indeed, one study found that combining Pb-nitrate with soluble chromate was as potent as Pb-chromate and greater than soluble chromate alone (Elias et al., 1991).

Thus, all together, these studies suggest that Pb ions alone cannot transform rodent cells; however, they may be co-carcinogenic or promote the carcinogenicity of other compounds. These data are in contrast to findings described in the1986 Pb AQCD that included a positive study. One possible factor may be exposure duration; the study in question indicated that the Pb-transformed cells were exposed for 9 days. The studies discussed here all exposed cells for 7 days or less. Further careful study of a time course of exposure is necessary to determine whether Pb actually induces transformation in cultured rodent cells.

Nonmammalian Cell Cultures

No carcinogenesis studies were located that used nonmammalian cell culture models.

5.6.2.4 Organ-Specific Studies

No organ-specific or organ culture studies concerning Pb carcinogenesis were located.

5.6.2.5 Carcinogenesis Summary

It still remains difficult to conclude whether Pb is a human carcinogen. The assessment of the carcinogenicity of Pb through human epidemiological studies remains ambiguous.

By contrast, the studies confirm that Pb is an animal carcinogen and further extend our understanding of the mechanism to include a role for metallothionein. The cell culture data suggest that Pb can induce anchorage independence, but whether it can induce full neoplastic transformation of human cells is uncertain.

5.6.3 Genotoxicity Studies

The human genotoxicity studies are only briefly reviewed in this section. For a more detailed review, see Chapter 6 (Section 6.7) in this document.

5.6.3.1 Human Studies

Adults

A number of studies investigating the potential genotoxicity of Pb have been conducted in human populations. Endpoints considered include chromosome aberrations, sister chromatid exchanges (SCE), micronuclei formation, DNA strand breaks, and hypoxanthine guanine phosphoribosyl transferase (HPRT) mutations. In general, these studies were much more specific than the carcinogenesis studies, as correlations with blood-Pb levels could be made, other confounders could be ruled out, and the endpoints were more short-term.

The chromosome damage studies are ambiguous and contained some methodological flaws. Four studies were positive (Xupei et al., 1988; De at al., 1995; Bilban, 1998; Pinto et al., 2000), while two were negative (Anwar and Kamal, 1988; Rajah and Ahuja, 1996). Moreover, the four positive studies included two that could not rule out potential contributions from other genotoxic metals and one that found a correlation only at very high blood Pb levels (>52 μg/dL).

By contrast, the studies of micronucleus formation (Bilban, 1998; Vaglenov et al., 1998; Pinto et al., 2000; Palus et al., 2003; Minozzo et al., 2004), SCE (Xupei et al., 1988; Bilban, 1998; Pinto et al., 2000; Duydu et al., 2001; Palus et al., 2003), DNA strand breaks (Restrepo et al., 2000; Fracasso et al., 2002; Hengstler et al., 2003; Danadevi et al., 2003; Palus et al., 2003) all consistently found clear correlations between Pb and genotoxicity. It should be noted that there were two negative studies for SCE (Rajah and Ahuja, 1995, 1996), but both were by the same group and considered the same very small population of workers (only 5 Pb-exposed workers) and, thus, may not have had enough power to detect potential differences.

It is notable that one study found an interesting correlation of HPRT mutation rates and blood Pb levels from environmental Pb exposure in Belgian women (Van Larebeke et al., 2004). This study is the first and only one to consider Pb-induced mutations. Further research is needed to assess the validity of these data.

Thus, it appears from these studies that Pb is genotoxic to humans, although it may not induce substantial amounts of chromosome damage. This conclusion is consistent with the laboratory studies discussed below. For more in-depth consideration of the epidemiology studies see Chapter 6, Section 6.7.

Children

Two recent studies of Pb-induced genotoxicity in children have been published. One study of children living in a high Pb contamination area of Czechoslovakia found no increase in chromosome damage in white blood cells compared with children living in an area with lower Pb contamination (Smejkalova, 1990). Comparisons were not done with children living in an area with little or no Pb contamination. Measurements of blood Pb levels indicated a statistical difference in blood levels between the two groups but not necessarily a substantial, or biologically significant, difference between them. (Typically the control group levels were in the high 20's compared to the low 30's μg/dL in the exposed group). Thus, the possibility that each group was exposed to a Pb level that could induce a baseline level of damage cannot be ruled out and, thus, it cannot be conclusively stated that Pb was not clastogenic in this study.

The other study found an increase in Pb-induced strand breaks in white blood cells from children living in an area of Mexico with high Pb contamination compared to children living in an area with lower Pb contamination (Yanez et al., 2003). Blood Pb levels confirmed a difference in exposure to Pb, but urinary arsenic levels confirmed that these children were exposed to higher levels of arsenic, too; and, thus, it cannot be determined which chemical was responsible for the damage.

The possible genotoxicity of Pb for children is an important concern. However, there are simply too few data to draw definitive conclusions, and more research is needed. See Chapter 6 (Section 6.7) for more in-depth discussion of the epidemiology of Pb in human populations.

5.6.3.2 Laboratory Animal Studies

Fourteen studies evaluated the genotoxicity of Pb compounds in animal models. The majority of these studies focused on mice, and the Pb was administered by intraperitoneal (IP) or intravenous (IV) injection. Several endpoints were considered including chromosome aberrations, SCE, micronucleus formation, and DNA strand breaks. Overall, the results are ambiguous, due in part to study design and the various endpoints considered. These studies are summarized in Table AX5-6.4.

Lead compounds appear to be able to damage chromosomes, if only weakly. Two studies with well-performed analyses were positive (Fahmy, 1999; Aboul-Ela, 2002). The other positive studies observed that Pb could induce karyotypic arrangements, indicating a possible clastogenic response; however, these studies did not analyze very many cells (Chakraborty et al., 1987; Nayak et al., 1989a,b; Dhir et al., 1990, 1992a,b; Nehez et al., 2000). Some found chromosome damage, but it did not increase with dose (Chakraborty et al., 1987; Nayak et al., 1989a,b; Dhir et al., 1990). Altogether, the data do suggest some role for Pb in inducing chromosome damage, but it may be a weak effect.

Similarly, the data for micronuclei and DNA damage are ambiguous. One study found that Pb induced micronucleus formation in a dose-associated manner, but only considered two doses (Roy et al., 1992). The other study found that Pb induced micronucleus formation but not in a dose-dependent manner (Jagetia and Aruna, 1998). This difference may reflect the somewhat shorter exposure time in the second study.

One DNA damage study found that Pb nitrate could induce DNA strand breaks in the white blood cells of mice (Devi et al., 2000); however, the damage was not dose-dependent. Another found DNA damage in a number of organs, but only one dose was considered and the authors described the effect as weak (Valverde et al., 2002). In both studies, the highest doses caused less damage than the moderate- to low-doses. These data again suggest that Pb is only weakly causing damage.

By contrast, the results for SCE are consistently positive. The three studies that were positive found that SCEs were induced in a dose-dependent manner (Fahmy, 1999; Nayak et al., 1989a; Dhir et al., 1993).

The route of administration complicates the interpretation of all of these genetic studies. All of the studies, except for three chromosome damage studies, used injection-based exposures.

It is unknown if exposures that reflect more realistic scenarios (e.g., from drinking water) would cause any of these effects. Only one study of DNA strand breaks used a physiologically relevant exposure (inhalation).

Four studies exposed animals by gavage, which is still a somewhat artificial exposure. One was a DNA damage study that found weak activity (Devi et al., 2000). The other three considered chromosome damage (Aboul-Ela, 2002; Dhir et al., 1992b; Nehez et al., 2000). Two found a dose-response for a 24 h-exposure to Pb nitrate-induced chromosome aberrations in mice (Aboul-Ela, 2002; Dhir et al., 1992b). The other found that a 4-week exposure to Pb-acetate induced aneuploidy, but not chromosome aberrations, in rats (Nehez et al., 2000). It is difficult to reconcile these two studies, as they use different exposure times, chemicals, and species. More work is needed using relevant doses and exposure conditions to Pb compounds in multiple species to determine if Pb induces chromosome aberrations.

Some studies also tried to offset the effects of Pb with a variety of compounds. Potential modulators included fruit extract from *Phyllanthus emblica*, ascorbic acid, calcium, and iron (Aboul-Ela, 2002; Dhir et al., 1990, 1992a, 1993; Roy et al., 1992). Other studies sought to determine if coexposure to other toxicants would potentiate the effects of Pb (Dhir et al., 1992b; Nehez et al., 2000) and considered both zirconium and cypermethrin. The data indicated that the fruit extract could block the toxic effects of Pb, an effect which may, in part, be attributable to ascorbic acid, but that other components must also be involved, because ascorbic acid alone produced variable results. Iron also had an effect, but only if given just before, or with, the Pb compound; post treatments with iron had no effect. Calcium had a strong effect.

The effects with zirconium and cypermethrin are less clear. Both were reported to exacerbate the effects of Pb, but the effects for both are complicated by experimental design problems. For example, zirconium only exacerbated Pb's effects when given simultaneously and not when given 2 h before, or after, Pb. This seems rather unusual as the total exposure to each was 24 h and, thus, simultaneous exposure occurred in every circumstance. Thus, the data would seem to suggest that a 22-h coexposure had no effect, but that a 24-h exposure did. Alternatively, there may have been some interaction of the two chemicals in the gut during coexposure, creating a more toxic species.

Interpretation of the cypermethrin study is complicated by its design and the results. Only 20 metaphases were analyzed for each animal, instead of the recommended 100. In addition, the

statistical analyses were done relative to untreated controls and not to animals treated with Pb or cypermethrin alone. Careful inspection of the tables reveals that actual exposure to Pb plus cypermethrin induced less damage than that induced by Pb alone. Thus, the effects of them together appear to be less than additive. More work is needed to explore the meaning of these data and the importance of Pb mixtures.

The previous report found a similar amount of ambiguity; some animal studies were positive for chromosome damage and others were negative. Other endpoints were not described after Pb exposure in experimental animals. These data suggest that Pb can induce SCE but that it can induce chromosome damage, DNA damage, or micronuclei either weakly or not at all.

5.6.3.3 Cell Culture Studies

Few cell culture studies were reported in the 1986 Pb AQCD. Since 1986, a great deal of theoretical and technological progress has allowed for a large number of cell culture studies to be performed, as discussed below.

Human Cell Culture

Mutagenicity

Two studies considered Pb-acetate-induced mutagenesis in human cells. Both considered mutations at the HPRT locus, with one using keratinocytes and the other skin fibroblasts (Ye, 1993; Hwua and Yang, 1998). These studies are summarized in Table AX5-6.5.

One study reported no lead-induced mutagenesis (Hwua and Yang, 1998) but sought to explore the importance of oxidative metabolism in lead-induced mutagenesis by co-treatment with 3-aminotriazole, a known catalase inhibitor. This co-treatment did not increase Pb-acetate-induced mutagenesis, suggesting that either catalase was not involved in this effect or that Pb is truly not mutagenic. It would be premature to conclude that oxidative metabolism is not involved in anchorage independence, as these are the only data and are limited to catalase. Further data is needed to elucidate whether oxidative metabolism is involved in this effect of Pb as well as further studies of lead-induced mutagenesis.

The other study reported that Pb-acetate induced mutagenesis (Ye, 1993). However, interpretation of this study is hampered by its methodology. The study did not actually measure HPRT mutations or colony formation, but rather it attempted a quicker methodology that

measured tritium incorporation. Although a shorter assay is highly desirable, the study did not verify the observed effects with standard methods, and, thus, it is uncertain if the tritium incorporation actually reflected lead-induced mutations.

One study considered Pb-chromate and found that it was not mutagenic (Biedermann and Landolph, 1990).

There are insufficient data at this point to conclude whether Pb is mutagenic in human cells, although the few data that exist are largely negative.

Clastogenicity

Ten studies investigated the ability of Pb compounds to induce chromosome damage in cultured human cells. All but one were essentially from the same research group, and all but two considered Pb-chromate. All were done using normal, or nearly normal, human cells. These studies are summarized in Table AX5-6.6.

Only two of those studies focused on the clastogenicity of Pb itself (Wise et al., 2004b, 2005), the remainder used Pb compounds but focused on either chromate or radioactive particles as the clastogenic species. These studies found that Pb-glutamate was not clastogenic.

All of the Pb-chromate studies found that Pb-chromate induced chromosome damage in a concentration-dependent manner. However, the effects were either attributed or demonstrated to be caused by chromate ions. Lead ions were produced by Pb-chromate, but they were not clastogenic.

There was one study of radioactive Pb (Martins et al., 1993). The focus was on the clastogenic activity of alpha particles, and the identity of the specific Pb salt was not provided. The alpha particles were able to induce chromosome damage.

Overall, the data appear to indicate that Pb does not induce chromosome damage in human cells, although more investigation of different compounds is needed.

DNA Damage

Studies of DNA damage in cultured human cells have considered DNA strand breaks, Pb-DNA adducts, and DNA-protein crosslinks for a variety of Pb compounds. The only clear positive damage induced by Pb was Pb-DNA adducts following Pb-chromate exposure, although the authors referred to them as Pb associated with DNA (Singh et al., 1999). It is uncertain if

these represent actual adducts or some weaker association. Two studies found no DNA strand breaks induced by Pb (Hartwig et al., 1990; Snyder and Lachmann, 1989), and one study involving several laboratories found no DNA-protein crosslinks after Pb exposure (Costa et al., 1996). The other study found DNA double-strand breaks, but these were attributed to chromate and not Pb (Xie et al., 2005). These studies are summarized in Table AX5-6.7.

One other study was positive (Wozniak and Blasiak, 2003), but the results were unusual and their impact uncertain. Specifically, this study found that Pb-acetate induced DNA single-strand breaks but that the amount of damage decreased with concentration, and ultimately the highest concentration had less damage than the control. DNA double-strand breaks were observed, but were lowest at the highest concentration. DNA-protein crosslinks were seen only at the highest concentration, and the authors attempted to explain the decrease in strand breaks with this effect. This explanation may partially correct, but it does not entirely explain the decreased amount of damage at the middle concentration. These data need to be repeated by an independent group before they can be fully assessed.

Together, these data suggest that Pb likely does not induce DNA damage; however, the data are still too limited to allow any definitive conclusions.

Human Cell Genotoxicity Summary

The cumulative data suggest that Pb is not mutagenic and does not induce chromosome aberrations or DNA damage in cultured human cells. It is interesting to note that Pb-induced SCEs have not been considered in human cells.

5.6.3.4 Animal Cell Cultures

Mutagenicity

The potential mutagenicity of Pb compounds in rodent cells was considered in six studies. In particular, three mutagenesis systems were considered: mutagenesis at the HPRT locus, the gpt locus, and mutations in sodium-potassium ATPase. The results are highly variable and may be specific to the Pb compound considered in each case. In particular, Pb-chromate and Pb-acetate appear to be nonmutagenic. Lead acetate was positive but only at highly cytotoxic concentrations. By contrast, Pb-chloride and Pb-sulfate appeared to be mutagenic at relatively nontoxic concentrations. These studies are summarized in Table AX5-6.8.

Insufficient data exist at this point to conclude whether or not Pb is mutagenic in animal cells.

Clastogenicity

Seven studies investigated the ability of Pb compounds to induce chromosome aberrations in cultured mammalian cells (Table AX5-6.9). Four of these studies considered Pb-chromate and further investigation revealed that chromate was responsible for the clastogenic effect (Wise et al., 1992, 1993; Blankenship et al., 1997). Three of these studies considered other lead-containing compounds (Wise et al., 1994; Lin et al., 1994; Cai and Arenaz, 1998). All but one were negative and that one only found a small response at a single high dose (Wise et al., 1994). Lower doses had no effect. Considered together, the studies indicate that Pb does not induce chromosomal aberrations in cultured mammalian cells.

Only two studies considered Pb-induced micronuclei in cultured mammalian cells. One was negative (Lin et al., 1994) and the other positive (Bonacker et al., 2005).

Four studies considered Pb-induced SCE in cultured mammalian cells. The results were predominately negative (three studies [Hartwig et al, 1990; Lin et al., 1994; Zelikoff et al., 1988]). Interpreting these studies, however, is complicated by the fact that too few metaphase cells (less than 30 per concentration) were analyzed in each study. The one positive study considered 100 metaphases per concentration, making those data more reliable (Cai and Arenaz, 1998).

DNA Damage

Several measures of DNA damage in cultured human cells have been investigated, including DNA single-strand breaks and DNA-protein crosslinks. Most Pb compounds did not induce DNA single-strand breaks. The exception was Pb-chromate, which did induce DNA strand breaks, but this effect was likely a result of the chromate ion. These studies are summarized in Table AX5-6.10.

Both Pb-chromate and Pb-nitrate induced DNA-protein crosslinks in cultured mammalian cells. These data suggest that Pb is genotoxic in this manner; however, it is thought that the Pb-chromate-induced DNA-protein crosslinks result from the chromate and that the method used for

Pb-nitrate is not sufficiently rigorous. Thus, while the data are certainly suggestive, they are insufficient to make any definitive conclusion.

Nonmammalian Cell Cultures

Only one study was located considering Pb in a nonmammalian model (Table AX5-6.11). This study found that Pb-chromate was not mutagenic in a bacterial assay. The compound was studied because of its chromate content and, given that it is the lone study, no definitive conclusions can be reached.

5.6.3.5 Cell-Free Studies

No cell-free studies concerning Pb carcinogenesis or genotoxicity were located.

5.6.3.6 Organ-Specific Studies

One study (Valverde et al., 2002) considered organ-specific effects (see Table AX5-6.4). That study found a different pattern of DNA strand breaks in mice after inhalational exposure to Pb-acetate. DNA in the brain and lung were damaged the most, kidney and liver next, then nasal epithelia and leukocytes, with no damage in testicle DNA. These data are intriguing, as they suggest organ-specific responses after a physiologically relevant exposure (inhalation). More research is needed, however, to fully assess the impact of these findings. Moreover, while the damage was statistically significant, the authors described the effects as weak.

5.6.3.7 Genotoxicity Section Summary

There is some ambiguity in the genotoxicity results, as some endpoints were positive while most were negative. Consistent with the animal study data, Pb can induce SCE in rodent cells, but it is unknown if it can do so in human cells because this has not been tested. Lead also seems to induce DNA-protein crosslinks in rodent cells.

5.6.4 Genotoxicity as it Pertains to Potential Developmental Effects

The human genotoxicity studies are only briefly reviewed in this section. For a more detailed review, see Chapter 6 (Section 6.7). Only limited animal data and no cell culture studies focused on this issue as a concern. The available data are described below.

Adults

One study was located that considered the effects of Pb on sperm quality and quantity. This study considered Pb, cadmium, and selenium levels in 56 nonsmoking volunteers (Xu et al., 2003). No effects on sperm quality were correlated with Pb exposure up to 10 μg/L.

Two studies were located on the effects of Pb on sperm morphology in animals (Fahmy, 1999; Aboul-Ela, 2002). Both were positive, indicating that Pb may have an effect on sperm. They also found that Pb induced DNA damage in the sperm (See Table AX5-6.4). These studies are summarized in Table AX5-6.12.

Children

No studies were analyzed that considered the genotoxic effects of Pb in children as a developmental hazard. There are two studies that considered the genotoxic effects of Pb in children. They were discussed in Section 5.6.3.1.

Three studies were located on the fetal effects of Pb-nitrate on the fetus (Kristensen et al., 1993; Nayak et al., 1989a,b). Lead induced an increase in resorptions and there were hints of possible fetal chromosome damage, but the methods were poorly described and much more work is needed before conclusions can be drawn. These studies are summarized in Table AX5-6.13.

5.6.5 Epigenetic Effects and Mixture Interactions

Lead has been proposed to be a co-mutagen or possibly a promoter. Thus a number of epigenetic mechanisms have been proposed. Epigenetic effects occur when a compound such as Pb induces changes in cellular processes that do not result from changes in DNA sequence. In other words, Pb has been proposed to alter cells in ways that may change the cell without breaking or mutating DNA. There are three possible mechanisms: (1) alterations of gene expression that can stimulate cells to grow (mitogenesis) and/or can interfere with DNA repair; (this possibility has been investigated in several studies); (2) interaction with other metals; and (3) alteration of oxidative metabolism. Neither of the latter two have been extensively investigated.

5.6.5.1 Gene Expression

It has been argued that Pb may induce or co-induce carcinogenesis by altering cellular metabolism or by altering the metabolism of another chemical. Both whole animal and cell culture studies have been conducted to address this question and are described below.

Animal

Animal studies indicate that Pb can induce the expression of some phase I metabolizing enzymes, such as cytochrome P4501A1, and phase II metabolizing enzymes, such as glutathione and glutathione-S-transferase. These studies are summarized in Table AX5-6.14.

Thus, it is plausible that through this mechanism, Pb may act as a co-carcinogen by affecting the metabolism of other chemicals or possibly as a direct carcinogen by enhancing endogenously-induced damage. However, no studies have directly shown that such Pb effects are linked to cancer or alter the potency of another chemical; and, thus, it remains only a plausible hypothesis.

Human Cell Culture Studies

A few human cell culture studies have been done, and these generally confirm the animal studies. These studies are summarized in Table AX5-6.15.

Lead has been shown to affect the induction of some phase I metabolizing enzymes (such as cytochrome P4501A1) and phase II metabolizing enzymes (such as glutathione and glutathione-S-transferase and NAPDH oxidase). These experiments also indicate that Pb can affect the metabolism of other carcinogenic compounds, although they do not show that the genotoxic or carcinogenic effects change as a result of these effects; and, thus, more work remains to make this more than just a plausible explanation.

Animal Cell Culture Studies

No animal cell culture studies concerning the effects of Pb on the expression of metabolic genes were located.

5.6.5.2 DNA Repair

It has been argued that Pb may induce or co-induce carcinogenesis by altering the repair of DNA lesions induced by another agent. The greatest focus has been on damage induced by ultraviolet (UV) light. Only cell culture and cell-free studies have been conducted to address this question and are described below.

Human

Only one study considered Pb-induced effects on DNA repair in cultured human cells (see Table AX5-6.16). This study found that coexposure to Pb caused persistence of strand breaks induced by UV light. This persistence suggests that Pb interfered with the repair of these lesions, but direct evidence of that interference was not provided. These are the only data in human cells and, thus, it cannot be determined if Pb inhibits DNA repair in human cells.

Mammalian Cell Culture Models

Two studies considered Pb-induced effects on DNA repair in cultured mammalian cells. These studies are summarized in Table AX5-6.17. Both found that Pb-acetate increased UV-induced DNA damage including SCE, mutagenesis, and cytotoxicity. Lead did not affect strand breaks induced by UV. These data suggest that Pb may indeed inhibit repair, although direct interactions with repair proteins were not demonstrated.

Cell Free Systems

One study considered the effects of Pb on DNA repair proteins (McNeill et al., 2004). That study found that Pb can inhibit APE nuclease in cell-free systems.

5.6.5.3 Mitogenesis

It has been argued that Pb may induce or co-induce carcinogenesis by inducing cells to grow when they should not. Both animal and cell culture studies have been conducted to address this question and are described below.

5.6.5.3.1 Animal

Several studies have considered Pb-induced mitogenesis in animal models. These studies

are summarized in Table AX5-6.18. These studies found that Pb can stimulate cell growth, but primarily in the liver. One study did consider TNF-α expression in brain cells, but it was not demonstrated whether these effects were mitogenic. The interpretation of many of the studies is complicated by the exposure method (IV injection), which does not reflect human exposure. In general, the data indicate that Pb is mitogenic to the liver.

Human Cell Culture Studies

A number of studies have considered the potential growth-stimulatory effects of Pb in cultured human cells (Table AX5-6.19). These studies all found that Pb did not stimulate cell growth. Thus, mitogenesis is not a likely epigenetic effect for Pb in human cells.

Mammalian Cell Culture Studies

A number of studies have considered the potential growth-stimulatory effects of Pb in cultured mammalian cells other than the kidney. These studies all found that Pb did not stimulate cell growth. Thus, mitogenesis is not a likely epigenetic effect of Pb in human cells. One study found an increased mitotic index; however, it did not consider possible cell cycle arrest (Lin et al., 1994). Indeed, another study found that Pb increased the mitotic index, because it induced M-phase arrest (Wise et al., 2005).

Other

Lead-induced oxidative damage has been investigated as a potential cause of genotoxic or carcinogenic effects. Generally, the results suggest that Pb only produces low levels of reactive oxygen species, but that it may inhibit some enzymes involved in oxidative metabolism (Table AX5-6.20). Thus, Pb may affect oxidative metabolism, but more work is needed to draw meaningful conclusions.

5.6.5.4 Epigenetic Mechanisms Summary

The collective data support the hypothesis that Pb can induce an epigenetic effect. Lead can alter the expression of metabolic genes in cultured cells and may alter DNA repair, although much more study is needed. Lead may also affect oxidative metabolism or interact with other

metals, but again more study is needed. By contrast, it is unclear if Pb is mitogenic. It is mitogenic to the liver in animals, but it is not mitogenic in cultured cells. More study is needed to determine if this difference reflects differences between in vivo and cell culture models or if this property is specific to only certain organs, e.g., the liver.

5.6.6 Overall Conclusions

The overall conclusions have not changed much from the 1986 Pb AQCD. Lead remains an ambiguous carcinogen in humans and a clear carcinogen in animals. Cell culture studies support both of these conclusions, as effects in rodent cells were not seen in human cells. Lead does appear to be genotoxic in human epidemiology studies. By contrast, the laboratory studies are more ambiguous in both animal and cell culture studies. In these systems, the genotoxicity in culture is limited to SCE and, perhaps, to DNA-protein crosslinks. For other endpoints, it is only weakly active, if at all. Lead has not been evaluated sufficiently as a potential genotoxic hazard, but this probably stems from the fact it appears to be weakly genotoxic. The available data suggest that Pb can damage sperm and affect fetuses. More work is urgently needed on this topic. Cell culture studies do support a possible epigenetic mechanism or co-mutagenic effects.

5.7 LEAD AND THE KIDNEY

5.7.1 Review of Earlier Work

This section summarizes key finding from the 1986 Pb AQCD on the effects of Pb on the kidney in animals. Human studies published since 1986 are then reviewed in Section 6.4.

Both in vivo and in vitro studies on several different animal species revealed that renal accumulation of Pb is an efficient process that occurs in both proximal and distal portions of the nephron and at both luminal and basolateral membranes (Victery et al., 1979a; Vander et al., 1977). The transmembrane movement of Pb appears to be mediated by an uptake process that is subject to inhibition by several metabolic inhibitors and the acid-base status of the organism. Alkalosis increases Pb entry into tubule cells via both the luminal and basolateral membranes (Victery et al., 1979b).

Goyer et al. (1970a) were principally responsible for defining the role of renal proximal tubular nuclear inclusion bodies in the response to Pb intoxication. In addition to the early

reports of nuclear inclusion bodies appearing in the proximal tubule following Pb exposure (Goyer et al., 1970b), biochemical studies on the protein components of isolated rat kidney intranuclear inclusion bodies have shown that the main component has an approximate molecular weight of 27 kDa (Moore et al., 1973) or 32 kDa (Shelton and Egle, 1982) and is rich in glutamate and aspartate. Goyer et al. (1970c) suggested that the intranuclear inclusion body sequesters Pb, to some degree, away from sensitive renal organelles and metabolic pathways. Goyer et al. (1975, 1978) also showed that single or repeated administration of $CaNa_2EDTA$ leads to the disruption of the nuclear inclusion bodies and their removal from the nuclei. Rats treated for 24 weeks with both Pb and $CaNa_2EDTA$ had no inclusion bodies, but showed early interstitial nephropathy. As an extension of this study, Cramer et al. (1974) examined renal biopsies from 5 Pb workers with 0.5 to 20 years of exposure. The two workers with normal GFRs, and shortest exposure duration, showed intranuclear inclusion bodies, whereas the remaining three workers had no intranuclear inclusions but showed peritubular fibrosis.

Formation of intranuclear inclusion bodies was a common pathognomic feature for all species examined. In addition, proximal tubular cytomegaly and swollen mitochondria with increased numbers of cytosomes were also observed (Fowler et al., 1980; Spit et al., 1981). The morphological changes were principally localized in the straight (S3) segments of the proximal tubule. Goyer (1968) and Goyer et al. (1968) had demonstrated earlier that, after lead exposure, mitochondria were not only swollen but had decreased respiratory control ratios (RCRs) and inhibited state-3 respiration.

Aminoaciduria has been reported in several studies (Studnitz and Haeger-Aronson, 1962; Goyer et al., 1970b; Wapnir et al., 1979). Other studies have reported increased urinary excretion of electrolytes (e.g., sodium, potassium, calcium, water) following Pb administration (Mouw et al., 1978). Victery et al. (1981, 1982a,b, 1983) found that zinc excretion was increased following injection of lead.

Wapnir et al. (1979) observed that Pb-acetate administration caused a reduction in renal alkaline phosphatase activity and an increase in Mg-ATPase activity, but no significant changes in NaK-ATPase activity. On the other hand, Suketa et al. (1979) found marked a decrease in renal NaK-ATPase activity following a single oral administration of Pb-acetate at a dose of 200 mg/kg, but no change in Mg-ATPase.

Renal ALAD was found to be inhibited by Pb in both acute and chronic experiments (Silbergeld et al., 1982). Renal ALAD was similar to control levels when GSH was present but was significantly reduced in the absence of GSH (Gibson and Goldberg, 1970). Accumulation of both ALA and porphobilinogen was also observed in kidney tissue of Pb-treated rabbits, compared to controls. Other studies have not shown a reduction in renal ALAD following Pb exposure (e.g., Fowler et al., 1980). Higher levels of Pb may be required to cause the reduction in ALAD reported by Silbergeld et al. (1982), and it may possibly involve Pb-binding proteins in the kidney.

5.7.2 Markers of Renal Toxicity

The establishment and validation of new screening tests for nephrotoxic effects have been principally due to the efforts of the Belgian group (Price et al., 1996; Price, 2000; Lauwerys et al., 1992). They proposed the following battery of tests be used to screen both environmentally exposed and occupationally exposed individuals: (1) measures of glomerular integrity, i.e., urinary high-molecular weight proteins (albumin, IgG, transferrin); (2) measures of tubular absorption and secretion, i.e., low-molecular weight proteins (retinol binding protein, α-1-microglobulin; (3) measures of tubular integrity, i.e., enzymes, lysosomal N-acetyl β-D-glucosaminidase (NAG), brush border alanine aminopeptidase, brush border intestinal alkaline phosphatase, nonspecific alkaline phosphatase, α-glutathione-S-transferase (GST), and brush border antigens (BB50, BBA, HF5); (4) measures of glomerular and distal tubular function, i.e., prostanoids (thromboxane B2, prostaglandin F2 alpha, 6-keto prostaglandin F1alpha) ; (5) measures of glomerular structural proteins (fibronectin and laminin fragments); and (6) measures of distal tubular function, i.e., Tamm-Horsfall protein and π-GST. Other useful markers include urinary β_2-microglobulin, as a marker of proximal tubular integrity; PGE_2 and PGF_2, distal nephron markers; kallikrein, a marker of the distal tubule; lysozyme, ribonuclease, and γ-glutamyl transferrase, enzymes reflecting proximal tubule integrity; and sialic acid, an extracellular matrix marker (Fels et al., 1994; Pergande et al., 1994; Taylor et al., 1997). One or several of these urinary markers have been used in screening tests for human Pb workers and in animal studies of renal nephrotoxicity.

Questions have been raised about the usefulness of urinary NAG due to the absence of light or electron microscopic changes in low-dose Pb-treated animals who showed substantial

increases in NAG (vide infra) (Khalil-Maesh et al., 1993). Furthermore, Chia et al. (1994) found that urinary NAG in workers exposed to Pb correlated best with recent blood lead changes, suggesting that the increased urinary NAG activity reflected an acute response to a sharp increase in the renal Pb burden rather than to exocytosis. Questions have also been raised about the value of measuring the vasoconstricting prostariod cytokine thromboxane B2 (TXB_2) and the vasodilating prostanoid 6-keto prostaglandin F1 alpha (PGF1 alpha). Conflicting results have been reported in human Pb-exposed workers. Cardenas et al. (1993) reported an elevation in TXB_2 and a diminution in PGF1 alpha in 41 Pb-exposed workers in contrast to 41 controls. Hotter et al. (1995), on the other hand, reported that both substances were increased in 69 Pb-exposed workers in contrast to 62 controls. Blood Pb levels in the two worker groups were comparable, i.e., 48 µg/dL in the first group and 43 µg/dL in the second. In animal experiments (Gonick et al., 1998), the excretion of both prostanoids was equal in low-Pb (100 ppm)-fed rats as contrasted to normal controls after 3 months, despite an elevation in blood pressure in the Pb-fed rats. Blood Pb in the Pb-fed rats averaged 12.4 µg/dL compared to 1 µg/dL in the controls. Thus, measurements of these prostanoids remain of questionable value.

Attempts to validate nephrotoxic markers were conducted by Pergande et al. (1994), utilizing Pb-exposed workers as contrasted to normal controls. They found that about 30% of the Pb workers showed an increased excretion of α_1-microglobulin, NAG, ribonuclease, and/or Tamm-Horsfall protein, with positive correlations between these tubular indicators and blood Pb concentration.

5.7.3 Biochemical Mechanisms of Lead Toxicity

Nolan and Shaikh (1992) summarized what was known about biochemical mechanisms underlying Pb-induced toxicity at that time. A more detailed description based on recent animal studies follows in the next section.

The initial accumulation of absorbed Pb occurs primarily in the kidneys. This takes place mainly through glomerular filtration and subsequent reabsorption, and, to a small extent, through direct absorption from the blood. Lead may be taken up by the renal tubular epithelial cells from the basolateral side by active transport of the free ion. Smaller amounts can also cotransport with low molecular weight organic anions. The uptake of Pb through the renal brush border does not appear to occur via any specific carriers. Instead, the process may involve binding of Pb to

nonspecific surface sites on the brush border membrane, followed by internalization via endocytosis. Acute kidney damage due to Pb manifests primarily in the proximal tubules. The ultrastructural changes observed in acute experimental Pb nephropathy include both specific and nonspecific effects on the proximal tubular epithelium, e.g., dilation of the endoplasmic epithelium, blebbing of the nuclear membrane, enlargement of the autophagosomes, changes in mitochondrial structure, formation of inclusion bodies. Chronic exposure to Pb affects glomerular filtration, renal clearance, and tubular reabsorption and can lead to renal failure from interstitial nephritis.

Kidneys of chronically exposed individuals often show fewer or no nuclear inclusion bodies compared to kidneys of acutely exposed individuals. The specific ultrastructural changes associated with Pb nephropathy are the formation of cytoplasmic and nuclear Pb inclusion bodies (discussed at greater length below). These inclusion bodies are not limited to the proximal tubular epithelium, and have also been observed in peritoneum, astrocytes, neuroblastoma cells, and osteoclasts upon Pb exposure. The inclusion bodies are roughly spherical and typically consist of an electron-dense core, with a fibrillary network at the periphery. Research has revealed that the formation of the nuclear inclusion bodies is preceded by the synthesis of cytoplasmic inclusion bodies with a very similar structure. A protein unique to these structures is rich in acidic amino acids and has an isoelectric point of 6.3 and a molecular weight of 32 kDa. Two additional proteins with apparent molecular weights of 11.5 kDa and 63 kDa have been identified in kidney extracts. Both of these proteins have a high affinity, but little capacity, for binding lead. A Pb-binding protein of 12 kDa molecular weight was identified in the supernatant of brain homogenate from Pb-treated rats. A Pb binding protein of 10 kDa has also been isolated from the erythrocytes of Pb-exposed workers.

Mitochondrial function, in addition to structure, is very sensitive to lead. Changes include the uncoupling of oxidative phosphorylation, decreased substrate oxidation, and modification of ion transport processes. Other effects of Pb on cellular energetics include chelation of ATP and inhibition of microsomal NaK-ATPase. These changes may account for the proximal tubular dysfunction seen with acute Pb poisoning in children.

A new area of investigation of the mechanism of Pb toxicity was initially proposed by Quinlan et al. (1988) and Hermes-Lima et al. (1991). Both investigators proposed that free radicals, or ROS, stimulated by lead, may accelerate iron-dependent lipid peroxidation, causing

tissue injury. Hermes-Lima et al. (1991) stated further that ALA, which is formed in large amounts in Pb toxicity, may undergo enolization and autoxidation, yielding ROS. Autoxidation of ALA, in the presence or absence of iron complexes, yields superoxide, peroxide, and hydroxyl radicals. Gurer and Ercal (2000), based on several animal studies to be discussed below, have proposed that antioxidant supplementation following Pb exposure may provide a partial remedy by restoring the cell's antioxidant capacity.

5.7.4 Animal Studies

Two excellent review articles have been written about the effects of heavy metals on, and their handling by, the kidney (Barbier et al., 2005) as well as the mechanisms of kidney cell injury from metals (Fowler, 1992). The interested reader is directed to these reviews, although individual effects and mechanisms will be discussed subsequently.

5.7.4.1 Lead Toxicokinetics

deVries et al. (1998) published a model for Pb toxicokinetics to be used in planning treatment. The model is a four-compartment model with first-order kinetics. The four compartments of this model are blood, bone, liver, and kidney. Soft tissues are represented by the kidney and liver compartments. In addition, intake and excretion are included in the model. Excretion of Pb is mainly via the kidneys (70 to 80%), via bile and feces (15%), via nails, hair, and sweat (8%). The blood makes up the central compartment from which Pb is distributed after uptake in the body. The blood compartment contains about 4% of the total body burden of lead, and within this compartment, the Pb is mainly taken up by erythrocytes. The half-life of Pb in blood is about 30 days. From the blood, Pb is distributed relatively quickly to the soft tissues and bone. The distribution constant from blood to bone is much higher than the one from bone to blood, resulting in the accumulation of Pb in bone. The half-life in the soft tissues is about 30 to 40 days. Most of the body burden of Pb can be found in the bone compartment (~94%), where the half-life of Pb is several decades. Because of the vast amount of Pb in bone, a rebound in blood Pb usually occurs after chelation therapy. This model can be compared with a toxicokinetic model developed by Marcus (1985a,b,c) and further explored by Hogan et al. (1998), as discussed in Chapter 4 of this document.

Dieter et al. (1993) examined the effect of the nature of the Pb salt on the oral intake of Pb in male F344 rats. For 30 days, they administered doses of 0, 10, 30, and 100 ppm Pb in the form of soluble Pb-oxide, Pb-acetate, Pb-sulfide, and Pb-ore. At 100 ppm of Pb-acetate or soluble Pb-oxide, the rats developed ~80 µg/dL of blood and ~200 µg/g of bone Pb levels, whereas rats fed Pb-sulfide or Pb ore developed ~10 µg/dL of blood Pb and 10 µg/g of bone lead. In rats fed Pb-acetate or soluble Pb-oxide, blood Pb progressively increased with increasing dose, while in the other two groups measurable levels of Pb were observed only at the highest dose (100 ppm).

5.7.4.2 Pathology, Ultrastructural, and Functional Studies

Two important series of studies contrast the pathological and functional changes in the kidney after prolonged exposure to lead, with and without chelation therapy (i.e., DMSA or $CaNa_2EDTA$). In the first series of 3 long-term studies, Khalil-Manesh et al. (1992a,b, 1993a) described the effects of Pb-acetate on renal function and morphology in male Sprague-Dawley rats fed a low-calcium diet. Lead acetate was used in concentrations of 0.5% (high dose) and 0.01% (low dose) in drinking water for periods from 1 to 12 months, and then Pb-exposed animals were compared to pair-fed controls (12 rats in each group). In all studies GFR was measured as ^{125}I-iothalamate clearance by a single injection technique. Urinary markers included NAG, GST, and brush border antigens (BB50, HF5, and CG9) and were expressed as units/g creatinine. Blood and urine Pb were measured prior to sacrifice in each group of animals. Wet and dry weights of kidneys were determined, then the kidneys were processed for light, electron, and immunofluorescent microscopy.

In the first study (Khalil-Manesh et al., 1992a), animals treated with continuous high-dose Pb for 12 months reached a maximum blood Pb of 125.4 ± 10.1 µg/dL after 6 months, at which time the dose of Pb was reduced from 0.5% to 0.1%. Blood Pb at the end of 12 months averaged 55 µg/dL. Urine Pb remained above 100 µg/g creatinine at all times, but it was highest at 3 months, averaging 340 µg/g creatinine. In the Pb-treated animals, GFR was increased above controls at 3 months (1.00 ± 0.14 vs. 0.83 ± 0.26 mL/min/ 100 g body wt, $p = 0.05$), then declined after 6 months to 0.78 ± 0.16 vs. 0.96 ± 0.08 mL/min/100 g body wt in controls (Figures 5-7.1 and 5-7.2). As indicated by the ratio of kidney dry/wet weight, increased kidney tissue mass was observed during the first 3 months of Pb exposure, but decreased tissue mass

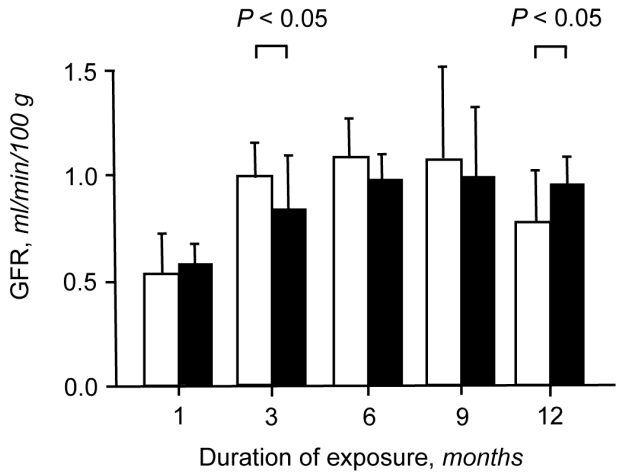

Figure 5-7.1. Changes in GFR of experimental high-dose lead and control animals with duration of exposure to lead. Open and closed bars represent GFR in experimental and control rats, respectively.

Source: Khalil-Manesh et al. (1992a), with permission.

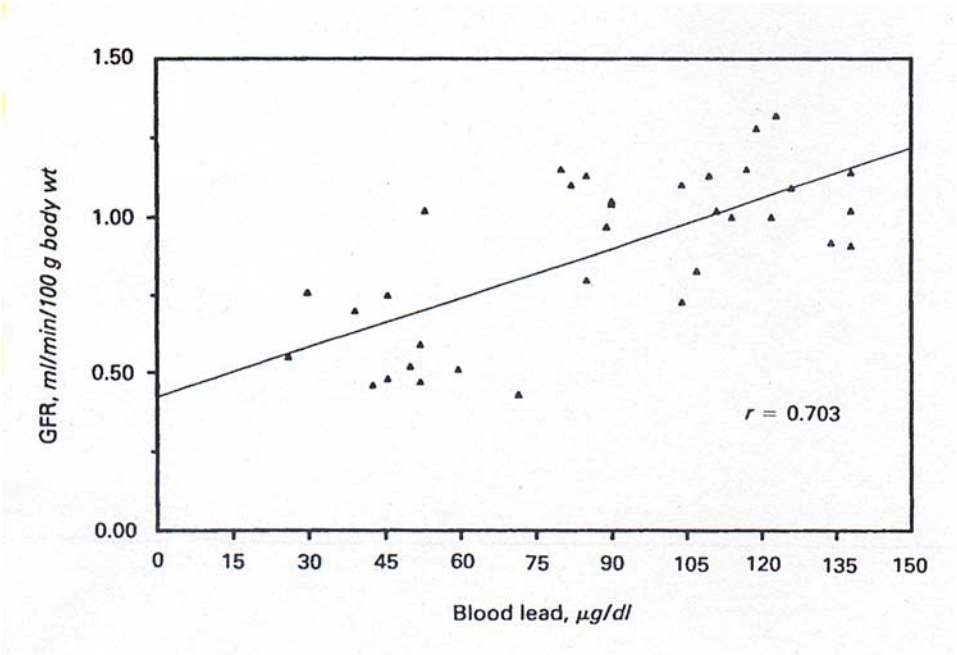

Figure 5-7.2. Correlation between GFR and blood lead during the first 6 months of high-dose lead exposure.

Source: Khalil-Manesh et al. (1992a), with permission.

was observed by 12 months. With regard to urinary markers, NAG was elevated above control levels at 3, 6, and 9 months of Pb exposure; GST was elevated at 3, 6, and 12 months of Pb exposure; and no significant differences were observed in the brush border antigens. Proximal tubular nuclear inclusion bodies were present at all time periods in Pb-treated animals. Enlargement of proximal tubular cells and nuclei were seen beginning at 3 months. At 6 months, focal tubular atrophy and interstitial fibrosis appeared, increasing in extent up to 12 months. Mitochondrial alterations, consisting of rounding and elongation, appeared by 1 month and were persistent. Glomeruli were normal through 9 months, but, at 12 months, they showed focal and segmental sclerosis. There were no electron-dense deposits and immunofluorescent studies were negative. Renal arteries and arterioles were normal at all time point examined.

The second study (Khalil-Manesh et al., 1992b) consisted of the discontinuation of both the high- and low-dose Pb exposure after 6 months, then treatment with three courses of DMSA or discontinuation of high-dose Pb alone after 1, 6, and 9 months of Pb feeding. Controls were pair-fed, exposed to Pb for 6 months, then removed from exposure for 6 months without receiving DMSA. Low-dose Pb-treated rats showed no significant pathologically with or without DMSA treatment but exhibited a significant increase in GFR after DMSA treatment (1.09 ± 0.19 vs. 0.88 ± 0.22 mL/min/100 g body weight; $P < 0.03$) (Figure 5-7.3). Urinary markers remained unchanged, and there were no structural alterations by light or electron microscopy. High-dose Pb-treated animals showed no functional or pathologic changes when Pb exposure was discontinued after 1 month. However, when the duration of exposure was 6 or 9 months, GFR was decreased and serum creatinine and urea nitrogen were increased compared to controls. Tubulointerstitial disease was severe. Administration of DMSA resulted in an improvement in GFR (Figure 5-7.3) and a decrease in albuminuria, together with a reduction in size and number of nuclear inclusion bodies in proximal tubules. However, tubulointerstitial scarring was only minimally reduced. In conclusion, except for a brief initial exposure, discontinuation of high-dose Pb exposure failed to reverse Pb-induced renal damage. Treatment with the chelator, DMSA, improved renal function but had less effect on pathologic alterations. Because GFR improved after DMSA treatment in both low- and high-dose Pb-treated animals, irrespective of the degree of pathologic alterations, it may be concluded that the DMSA effect is most likely mediated by hemodynamic changes.

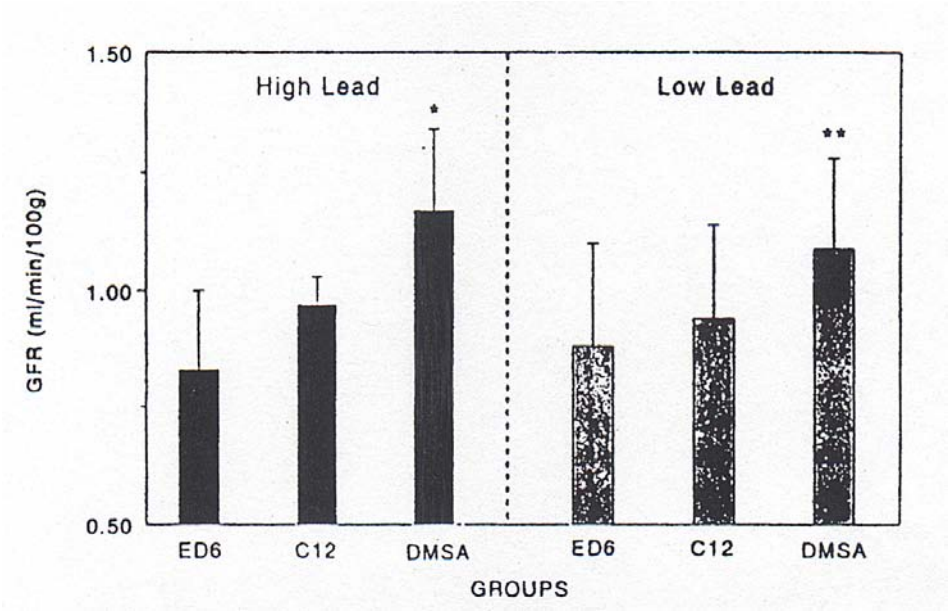

Figure 5-7.3. GFR in high-lead and low-lead experimental discontinuous (ED6) and DMSA-treated rats (DMSA) as compared to controls (C12). All rats were studied at 12 months.

*$p < 0.01$ when compared to ED6 and C12.
**$p < 0.05$ when compared to ED6.

Source: Khalil-Manesh et al. (1992b), with permission.

The third study (Khalil-Manesh et al., 1993a) examined the course of events over 12 months in continuous low level Pb-exposed animals. Maximum blood Pb levels in experimental animals were reached at 3 months, averaging 29.4 ± 4.1 µg/dL. GFR was found to be significantly increased above pair-fed controls at 1 and 3 months, but it was normal at other time periods (1 month experimental, 1.18 ± 0.12 vs. control, 0.76 ± 0.15mL/min/100 g; $p < 0.001$; 3 month experimental, 1.12 ± 0.16 , vs. control, 0.86 ± 0.10 mL/min/100 g; $p < 0.001$) (Figure 5-7.4). Levels of urinary NAG in Pb-exposed rats exceeded control levels at all time periods, except at 12 months, when the normal increase with aging obscured differences between experimental animals and controls (Figure 5-7.5). In contrast, urinary GST, a more specific marker of metal-associated proximal tubular injury, was normal at all time periods. Proximal tubular nuclear inclusion bodies were sparse and were observed only at 1 and 3 months.

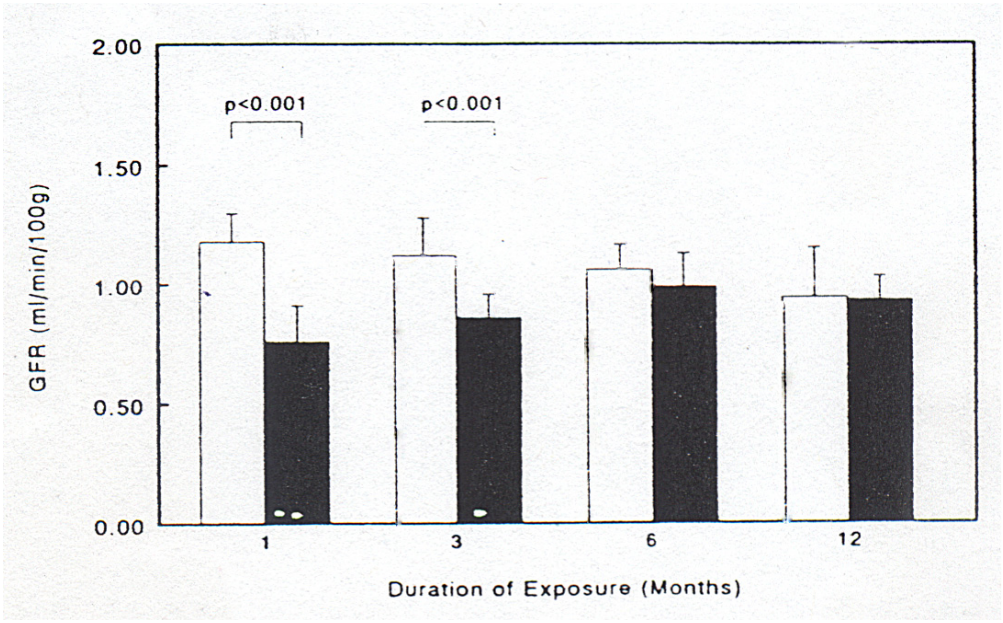

Figure 5-7.4. Changes in GFR in experimental and control rats, at various time periods.

Source: Khalil-Manesh et al. (1993a).

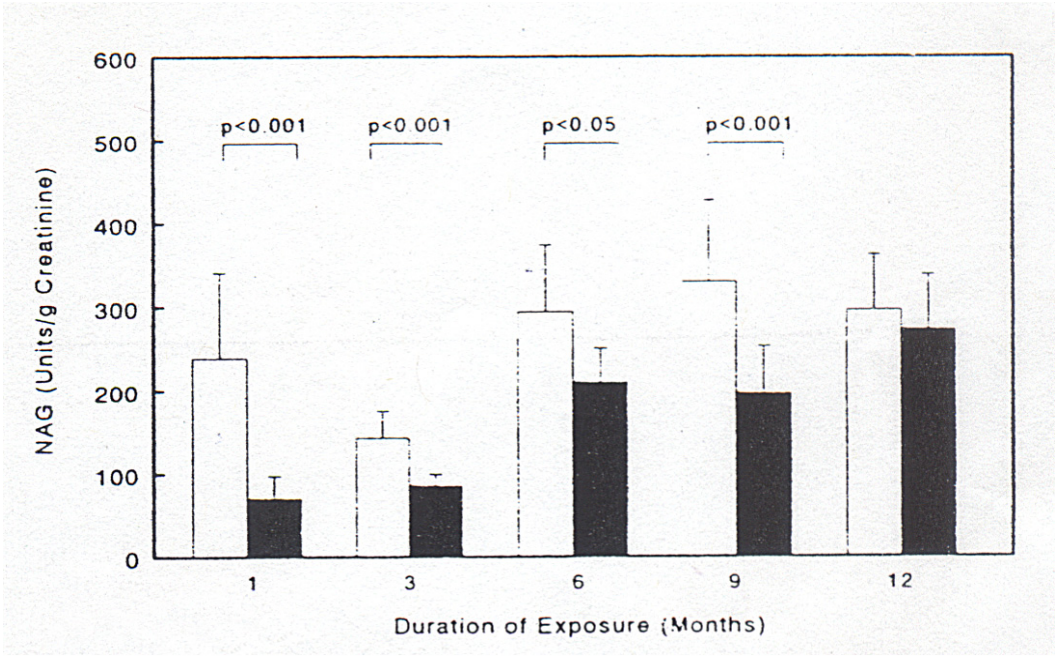

Figure 5-7.5. Urinary NAG concentration in experimental and control rats at various time periods.

Source: Khalil-Manesh et al. (1993a).

No other pathologic alterations were found in the kidneys until 12 months of exposure, when mild tubular atrophy and interstitial fibrosis were seen. The absence of changes in urinary GST accorded with the relative absence of morphologic changes, whereas the observed increases in urinary NAG suggest that this enzyme may be an overly sensitive indicator of tubular injury, more probably reflecting upregulation of the enzyme even in the absence of tubular injury. It should be noted that both low-dose Pb-treated animals and high-dose Pb-treated animals showed a "hyperfiltration" phenomenon during the first 3 months of Pb exposure. This observation could be invoked as a partial explanation for the late changes of glomerulosclerosis in the high-dose animals, but it cannot explain the lack of glomerular changes in the low-dose animals. Thus, these studies join those of Roels et al. (1994) and Hu (1991) in humans that indicate that Pb nephropathy should be added to diabetic nephropathy as diseases that lead to early hyperfiltration.

The second series of studies were performed by Sanchez-Fructuoso et al. (2002a,b). Sanchez-Fructuoso et al. (2002a,b) evaluated the effect of $CaNa_2EDTA$ on tissue mobilization of Pb in Wistar rats initially treated with 500 ppm Pb-acetate for 90 days, followed by treatment with three courses of $CaNa_2EDTA$ 50 mg/kg/day for 5 days, separated by 9 days, or placebo. Lead levels were measured in blood, urine, kidney, liver, brain, and femur. There was no change in bone Pb after $CaNa_2EDTA$ compared to placebo, but Pb levels were significantly reduced in all other tissues (Figure 5-7.6). The authors emphasized that there was no redistribution to brain. Cory-Slechta et al. (1987) had originally reported that with $CaNa_2EDTA$ chelation in rats Pb is preferentially mobilized from bone and then redistributed to other organs, including brain. The Sanchez-Fructuoso et al. (2002a,b) findings stand in contrast, explained by the authors as due to a 3-fold higher level of $CaNa_2EDTA$ used by Cory-Slechta et al. (1987).

Sanchez-Fructuoso et al. (2002b) also evaluated pathologic changes, as well as the response of ALAD activity before and after $CaNa_2EDTA$ treatment in the same rats. In the 90-day Pb-treated animals, the main findings were hypertrophy and vacuolization of medium and small arteries (Figure 5-7.7); mucoid edema and muscular hypertrophy in arterioles; loss of cell brush borders, cell loss, and intranuclear inclusion bodies in the proximal tubule; and fibrosis and the presence of infiltrates in the interstitial component. Treatment with $CaNa_2EDTA$ slowed the progression of most alterations (Figure 5-7.8) and resulted in a diminution in nuclear inclusion

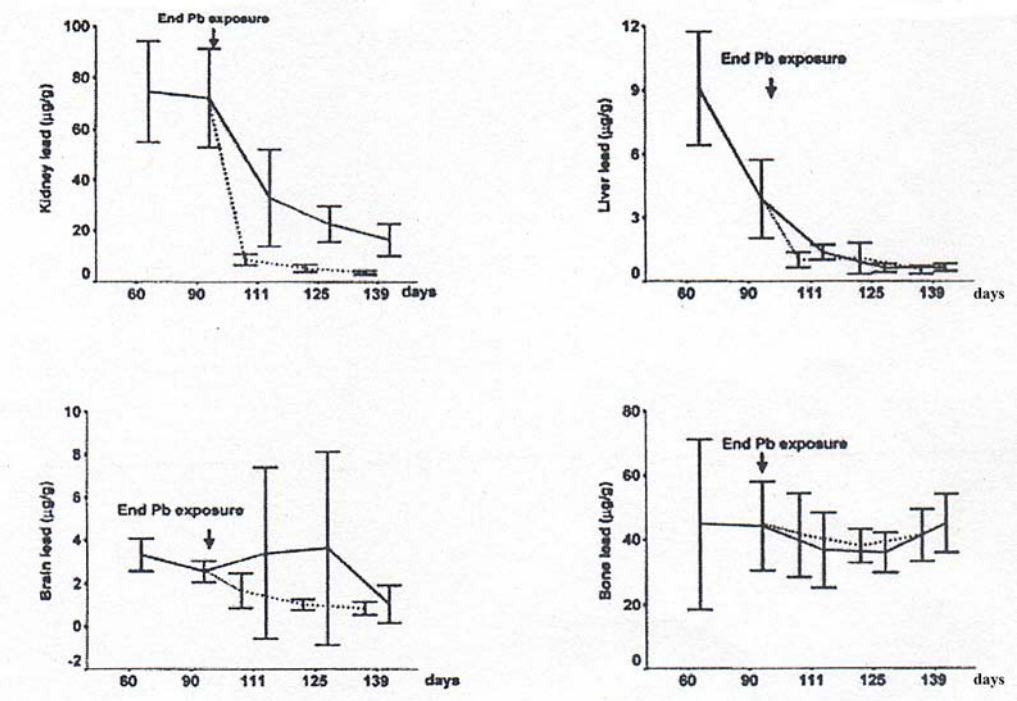

Figure 5-7.6. Kidney, liver, brain, and bone Pb levels in 56 Pb-exposed rats. After 90 days of poisoning, animals were administered serum saline (solid line) or calcium disodium EDTA (broken line).

Source: Sanchez-Fructuoso et al. (2002a), with permission.

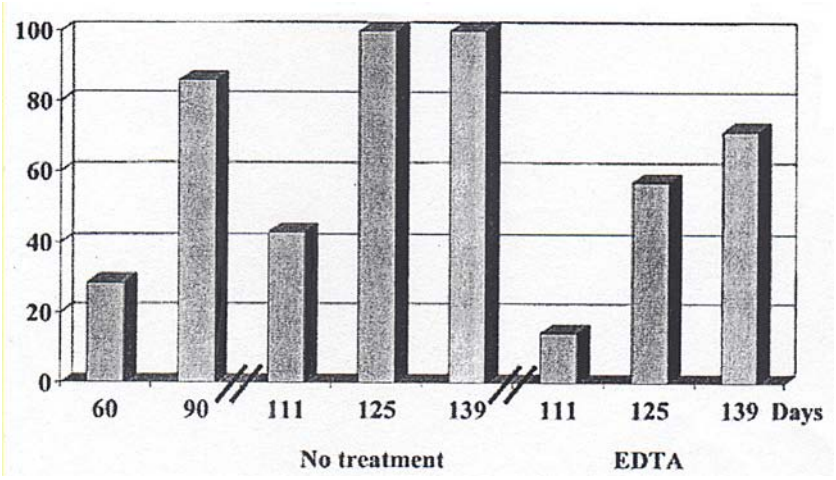

Figure 5-7.7. Percentage of moderate and severe hypertrophy and vacuolization lesions in small and medium sized arteries in the kidney of lead-exposed rats.

Source: Sanchez-Fructuoso et al. (2002b), with permission.

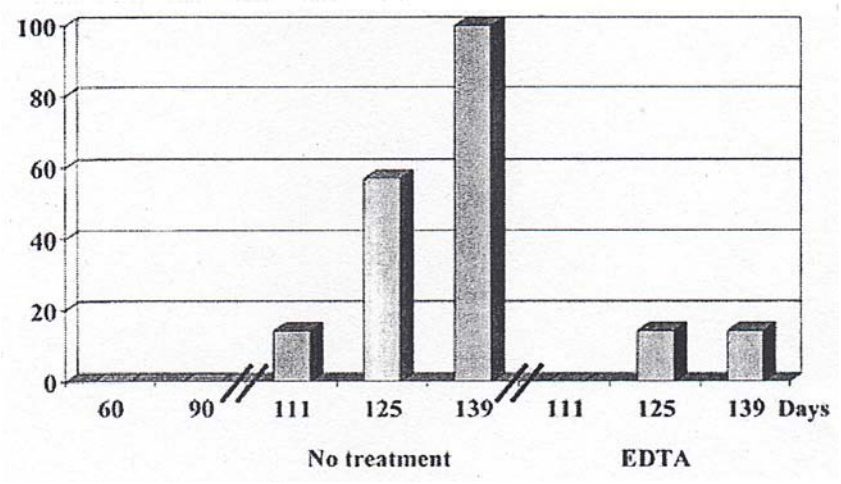

Figure 5-7.8. Percentage of moderate and severe muscular hypertrophy lesions in arterioles of the kidney in lead-exposed rats.

Source: Sanchez-Fructuoso et al. (2002b), with permission.

bodies. ALAD activity was reduced from 3.18 ± 0.52 U/mL in controls, to 0.82 ± 0.16 U/mL in the Pb-exposed rats. In the rats treated with $CaNa_2EDTA$, ALAD returned to near control levels (2.98 ± 0.41 U/mL) at 137 days. It is surprising that such remarkable vascular changes were noted in this study, while none were noted in Khalil-Manesh et al. (1992a), even with high-dose Pb for longer periods of time. The kidney content of Pb (mean 74.6 μg/g) was also lower than the mean kidney content at 12 months (294 μg/g) in the Khalil-Manesh et al. (1992a) study. The only explanation for these striking differences that can be offered is that different strains of rats were employed, i.e., Wistar in the Sanchez-Fructuoso (2002b) study and Sprague-Dawley in the Khalil-Manesh et al. (1992a) study. The presence or absence of hypertension cannot be invoked as an explanation, because in another Khalil-Manesh et al. (1993b) study the low-dose Pb animals became hypertensive while the high-dose animals did not. These and other related studies are summarized in Table AX5-7.1.

5.7.4.3 Biochemical Mechanisms of Lead Toxicity

Role of Free Radicals (Reactive Oxygen Species)

Since the early 1990s, it has been appreciated that free radicals, now known as reactive oxygen species (ROS), are involved in the manifestations of Pb poisoning, presumably via their

adverse effects on tissue integrity and/or their vasoconstrictive effects on vascular endothelium. Wolin (2000) produced an extensive review of individual ROS, and their interactions with NO, the major endogenous vasodilator, which acts via a second messenger, cGMP. The production of ROS often begins with a one-electron reduction of molecular oxygen to superoxide anion (O_2^-) by various oxidases. NAD(P)H oxidases are the principal enzymes involved. Superoxide anion is a negatively charged free radical that can be broken down to hydrogen peroxide (H_2O_2) by superoxide dismutase (SOD) or can interact with NO to form the highly reactive peroxynitrite ion ($ONOO^-$), which, because of its extremely short half-life, is measured as its reaction product, tissue nitrotyrosine. Catalase and glutathione (GSH) peroxidase (GSHRx) metabolize H_2O_2 to Compound I and oxidized glutathione (GSSG), respectively, while myeloperoxidase metabolizes H_2O_2 to hypochlorous acid (HOCl). The reaction of H_2O_2 with ferrous ion results in the formation of hydroxyl ion ($^{\bullet}OH$). ROS can be scavenged by endogenous thiols (e.g., GSH) or exogenous thiol, e.g., N-acetylcysteine (NAC). ROS can be measured as the concentration of the lipid peroxidation product, malondialdehyde-thiobarbituric acid (MDA-TBA) or by the more recently introduced F-2 isoprostanes.

Kumar and Das (1993) explored the involvement of ROS in the pathobiology of human essential hypertension. They found that plasma levels of lipid peroxides were higher in subjects with uncontrolled essential hypertension compared to normal controls. Angiotensin II, a potent vasoconstrictor, was found to stimulate free radical generation in normal leukocytes, which was thought to inactivate NO, and possibly prostacyclin, which can lead to increased peripheral vascular resistance and hypertension.

Hermes-Lima et al. (1991) also explored the involvement of ROS in Pb poisoning. They described the process of autoxidation of ALA in the presence or absence of iron complexes, which yields free radicals. Free radicals are also produced by Pb-stimulated iron-dependent lipid peroxidation, as determined by quantification of thiobarbituric acid-reactive species (TBARS). Pereira et al. (1992) demonstrated that chronically ALA-treated rats (40 mg/kg body weight every 2 days for 15 days) under swimming training reached fatigue significantly earlier than the control group, as well as demonstrating decreased mitochondrial enzymatic activities. In vivo prooxidant properties of ALA were also suggested by the observed increase of CuZnSOD in brain, muscle, and liver of untrained rats submitted to chronic treatment with ALA.

Ercal et al. (1996) contrasted the effects of treatment with DMSA or NAC in Pb-exposed C57BL/6 mice. Five weeks of Pb exposure was found to deplete GSH levels, increase GSSG, and promote MDA production in both liver and brain samples. Glutathione levels increased and GSSG and MDA levels decreased in groups of Pb-exposed mice that received 1 mmol/kg DMSA or 5.5 mM/kg NAC for 7 days prior to sacrifice. Treatment with DMSA caused reduction in blood, liver, and brain Pb levels consistent with its function as a chelating agent, while treatment with NAC did not reduce these Pb levels. However, NAC treatment reduced indices of oxidative stress in both brain and liver samples. Concentrations of blood Pb in controls were 0.5 ± 0.5 μg/dL; in Pb-treated mice, were 36.5 ± 2.4 μg/dL; in Pb + DMSA-treated mice, were 13.7 ± 1.3 μg/dL; and in Pb + NAC-treated mice, were 36.0 ± 3.5 μg/dL. Thus both DMSA and NAC acted as antioxidants, presumably via their thiol groups, but only DMSA reduced the concentration of lead.

Vaziri and co-workers (Gonick et al., 1997; Ding et al., 1998, 2000, 2001; Vaziri et al., 1997, 1999a,b, 2000, 2001a,b, 2003; Zhou et al., 2002; Ni et al., 2004) have published a number of articles relating to the production of ROS and alterations in enzymatic activities in Pb-induced hypertension. These were discussed in detail in Section 5.5 but are described briefly here. In the majority of studies, Pb-induced hypertension was produced by the administration of Pb-acetate, 100 ppm in drinking water, for 3 months to male Sprague-Dawley rats. Early studies (Gonick et al., 1997) revealed that hypertension could occur in the absence of changes in NO or cGMP but with an attendant rise in plasma and kidney MDA-TBA, indicating an increase in ROS. In a second study, Ding et al. (1998) showed that infusion of arginine, the precursor of NO, or DMSA, a thiol Pb chelator and antioxidant, reduced blood pressure to or towards normal, while simultaneously increasing depressed urinary NO and decreasing an elevated MDA-TBA. Ding et al. (2000, 2001) further showed that the ROS species, $^{\bullet}OH$, measured as salicylate-trapped 2,3 dihydroxybutyric acid, was increased in plasma and cultured rat aortic endothelial cells after exposure to lead, and that dimethylthiourea, a reputed scavenger of $^{\bullet}OH$, returned blood pressure, MDA-TBA, $^{\bullet}OH$, and nitrotyrosine to or towards normal. Ni et al., in 2004, demonstrated in both human coronary endothelial (EC) and vascular smooth muscle cells (VSMC) that Pb-acetate also increased superoxide (demonstrated by flow cytometry using hydroethidine) and H_2O_2 (demonstrated with dihydrorhodamine) production. After long-term (60-h) exposure, detectable superoxide levels fell to near normal while H_2O_2 production remained high.

Vaziri et al. (1997) showed that lazaroids, a class of non-thiol antioxidant, also restored blood pressure, NO, and MDA-TBA to normal. Vaziri et al. (1999a) studied rats treated for 12 weeks with either Pb-acetate alone or Pb-acetate + vitamin E-fortified food (5000 units/kg rat chow). They measured urinary excretions of stable NO metabolites (NO_x) and plasma and tissue abundance of nitrotyrosine, the footprint of NO oxidation by ROS. The Pb-treated group showed a marked rise in blood pressure; a significant increase in plasma and kidney, heart, liver, and brain nitrotyrosine abundance; and a substantial fall in urinary NO_x excretion. Concomitant administration of high-dose vitamin E ameliorated hypertension and normalized both urinary NO_x excretion and tissue nitrotyrosine without altering tissue Pb content. Vaziri et al. (1999b) also measured eNOS and iNOS in the aorta and kidney of Pb-treated and Pb + vitamin E-treated rats. Lead treatment increased both isotypes in aorta and kidney, signifying increased NO production, while Pb + vitamin E lowered aortic, but not kidney, expression of eNOS and iNOS. Vaziri and Ding (2001) tested the effect of lead, 1 ppm, on cultured human EC cells. Lead was tested alone or with either the SOD-mimetic agent, tempol, or a potent antioxidant lazaroid compound (both at 10^{-8} or 10^{-7}mol/L) on eNOS expression and NO production. Lead-treated cells showed a significant upregulation of endothelial eNOS, increase in protein abundance, and increase in the production of NO metabolites. Treatment with either tempol or lazaroids abrogated the Pb-induced upregulation of eNOS protein and NO_x production. Vaziri et al. (2001) also studied increases in NOS isoforms in vivo in Pb-induced hypertension and reversal by tempol. Both eNOS and iNOS were increased in kidney, aorta, and heart, while NOS was increased in cerebral cortex and brain stem, of Pb-treated rats; blood pressure and NOS isoforms were returned to normal by tempol. Vaziri et al. (2003) determined whether the oxidative stress in animals with Pb-induced hypertension is associated with dysregulation of the main antioxidant enzymes (i.e., SOD, catalase, and GSHPx), or increases in the superoxide-producing enzyme NAD(P)H oxidase. At the conclusion of the experiment, immunodetectable CuZnSOD, MnSOD, catalase, GSHPx, and the gp^{91}phox subunit of NAD(P)H oxidase were measured by Western analysis in the kidney, brain, and left ventricle of control and Pb-exposed rats. Lead exposure resulted in a significant increase in kidney and brain CuZnSOD with a significant increase in brain, and insignificant increase in kidney and heart, gp^{91}phox. In contrast, MnSOD, catalase, and GSHPx in the kidney, brain, and left ventricle were unchanged. Incubation with Pb-acetate did not alter SOD activity in vitro. Thus, animals with Pb-induced hypertension

exhibited oxidative stress, which was associated with mild upregulation of the superoxide-generating enzyme NAD(P)H oxidase, with no evidence of quantitative SOD, catalase, or GSHPx deficiencies.

Vaziri et al. (2000) demonstrated that induction of oxidative stress in normal animals (by feeding the GSH synthase inhibitor, buthionine sulfoximine, 30 mmol/L in drinking water for 2 weeks) led to an increase in blood pressure, a reduction of urinary NO_x, a 3-fold decrease in liver GSH, and an increase in nitrotyrosine in kidney, aorta, heart, liver and plasma. Administration of vitamin E + ascorbic acid ameliorated hypertension and mitigated nitrotyrosine accumulation despite persistent GSH depletion. This experiment demonstrated the importance of GSH in protecting against the adverse effects of ROS accumulation in normal animals. Majority of the studies reported by Vaziri and co-workers indicated that low Pb exposure induced hypertension to be primarily mediated by ROS-induced depletion of NO. NO production, on the other hand, is stimulated, as shown by the increase in eNOS and iNOS. Enzymatic control of ROS levels by low Pb is achieved by upregulation of NAD(P)H oxidase with no decrease in SOD, catalase, or GSHPx, i.e., the enzymes that breakdown ROS. Scavengers of ROS ameliorate the elevated blood pressure, while the depletion of the endogenous methyl scavenger, GSH, increases blood pressure in normal animals. No studies have been done to date to address the question of why high-dose Pb administration does not lead to hypertension.

Farmand et al. (2005) pursued enzymatic studies by activity measurements and measures of protein abundance in the rat kidney and aorta following the protocol of Gonick et al. (1997) whereby rats are fed Pb-acetate 100 ppm for 12 weeks. They demonstrated that the activities of CuZnSOD and catalase were increased by Pb administration in renal cortex and medulla, whereas GSHPx was unchanged. In the thoracic aorta, Pb exposure resulted in significant upregulation of CuZnSOD activity, while catalase and GSHPx activities were unchanged, CuZnSOD, MnSOD, and catalase protein abundance were likewise unchanged. However, guanylate cyclase protein abundance in the thoracic aorta was decreased. The authors suggested that the Pb-induced compensatory upregulation of CuZnSOD and catalase and the decrease in aortic guanylate cyclase may be related to Pb-induced hypertension.

Gurer et al. (1999a) evaluated whether captopril, an ACE inhibitor, acted as an antioxidant in Pb-exposed F344 rats. Lead acetate was given in drinking water for 6 weeks.

Group I were the controls; group II received 1100 ppm Pb for 5 weeks and plain water during the week 6; group III received 1100 ppm Pb for 5 weeks and, during the week 6, received water containing captopril (10 mg/day). Blood Pb concentrations in the control group measured 0.8 µg/dL; in the Pb treated group, 24.6 ± 20 µg/dL; in the Pb + captopril group, 23.8 ± 1.6 µg/dL. MDA concentrations in liver, brain, and kidney were increased by Pb administration and reduced to or towards normal by the Pb + captopril treatment. GSH concentrations were decreased by Pb administration and restored by Pb + captopril treatment, whereas GSSG concentrations were increased by Pb administration and reduced by Pb + captopril treatment. Thus, this study showed that captopril was capable of augmenting the reducing capacity of the cells by increasing GSH/GSSG ratios without affecting blood Pb concentrations.

McGowan and Donaldson (1987) examined total nonprotein sulfhydryl and GSH concentrations in liver and kidney as well as GSH-related free amino acid concentrations in liver, kidney, and plasma in 3-week-old Pb-treated (2000 ppm dietary lead) chicks. Cysteine, converted from methionine, is the rate-limiting amino acid in GSH formation. The availability of glutamate, cysteine, and glycine becomes important in the restoration of depleted GSH. GSH, nonprotein sulfhydryl groups, glycine, and methionine were increased versus controls in the liver, but only nonprotein sulfhydryl, glycine, cysteine, and cystathionine increased in the kidney. Plasma levels of cysteine, taurine, and cystathione were reduced. Thus, Pb, for short periods of time, increases GSH turnover. These and other studies are summarized in Table AX5-7.2.

Effect of Lead on Selective Renal Enzyme Levels

Effects of Lead on Renal NAG

Dehpour et al. (1999) studied NAG release by the rat kidney perfused with Pb-acetate at 10, 20, and 50 µg/dL for 120 min, or Pb + arginine (the substrate for NO), or Pb + L-NAME (an inhibitor of NOS). Lead acetate caused a time and concentration-dependent increase in enzymuria. Addition of arginine decreased, while addition of L-NAME increased, Pb-induced NAG release. Histologic studies showed damage to some of the proximal tubule epithelial cells in rats treated with 50 µg/dL Pb-acetate, damage that which was increased further by the addition of L-NAME.

Effect of Lead on Renal GST

Two studies (Moser et al., 1995; Oberley et al., 1995) reported the effects of Pb administration on GST isoforms in developing rat kidney. In the first study (Moser et al., 1995), rats were treated either acutely (14- and 50-day old rats given three daily injections of Pb-acetate, 114mg/kg) or chronically (Pb levels of 0, 50, 250, and 500 ppm in drinking water for 1, 2, 3, 4, and 7 weeks postnatal). Chronic treatment rats were also given a 0.66% low calcium diet or standard rat chow. Essentially all kidney cytosolic GSTs (Yb1, Yb2, Yp, Yc1, Yl, Yb3, Ya1, Ya2, Yk) increased in the acute experiment (1.1- to 6.0-fold). In the chronic experiment, all but one isoform (Yb3) increased, and these results were markedly exacerbated by placing the rats on a low-calcium diet (Yb1 and Yp increased >25-fold). In the second study (Oberley et al., 1995), pregnant rats were given 250 ppm Pb from conception until weaning, then pups received 500 ppm from weaning until termination at either 3 or 7 weeks of age. By 7 weeks, proximal tubular cells showed intranuclear inclusions, tubular injury, and interstitial fibrosis. Creatinine clearances were reduced (0.55 + 0.05 versus 1.05 + 0.07 mL/min/100g; $P < 0.001$). Treatment with Pb also caused large increases in the immunoreactive protein of Yc, Yk, Yb1, and Yp GST subunits in proximal tubules but did not increase in the antioxidant enzymes CuZnSOD, catalase, and GSHPx.

Another experiment that examined the effect of an acute dose of Pb as Pb-nitrate (100 µmol/kg IV) on GST levels in rat liver and kidney was reported by Planas-Bohne and Elizade (1992). Seventy hours after injection, there was a marked increase in GST activity in both organs, accompanied by induction of the isoenzyme GST 7-7 in the liver.

The relationship between GST induction by acute exposure to Pb-acetate and oxidative stress was explored by Daggett et al. (1998). Rats in the 72-h and 7-day experimental groups received three consecutive daily injections of 114 mg/kg body weight of Pb-acetate. The level of kidney GST was increased at 3, 6, 12, and 24 h after injection, but MDA levels remained unchanged. Immunohistochemical markers of oxidative stress and NO production (MnSOD, eNOS, iNOS, and 4-hydroxy-2-nonenal) also did not change. The authors concluded that the GST changes were not the result of oxidative stress.

Witzman et al. (1998) and Kanitz et al. (1999) utilized two-dimensional (2-D) gel electrophoresis to explore protein markers of Pb exposure. Witzman et al. (1998) gave three consecutive IP injections of Pb-acetate (114 mg/kg) to Sprague-Dawley rats, sacrificed them on

the fourth day, and subjected the cytosolic fraction of kidney homogenate to 2-D gel electrophoresis. Lead exposure caused detectable inductions in both GSTP1 and GSTM1 and caused quantifiable charge modifications in GSTP1. Kanitz et al. (1999) examined kidney protein expression in male rabbits injected with Pb-acetate (260, 360, or 100 μg/kg) designed to produce blood levels of 20, 40, or 80 μg/dL. Injections were given during weeks 6 to 10, followed by maintenance doses during study weeks 11 to 20. Kidney homogenates were subjected to 2-D electrophoresis. Significant quantitative changes occurred in 12 proteins in a dose-related manner. Four proteins cross-reacted with anti-rat GSTp1 (π-GST). Thus, both studies confirmed GST induction by lead.

Daggett et al. (1997) examined the effects of triethyl Pb administration on the expression on GST isoenzymes and quinone reductase in rat kidney and liver. Fischer 344 rats were given one IP injection of triethyl Pb chloride (10 mg/kg body weight) and subsequent changes in enzyme expression were measured. There was a significant increase in GST activity in kidney; all GST subunits were significantly elevated, the largest increase being a 3.2-fold increase in GST Yb1. In the liver, injection of triethyl Pb-chloride resulted in decreased GST activity. The largest decrease in subunits was a 40% reduction in GST Ya1. The activity of quinone reductase was elevated 1.5-fold in kidney and 2.7-fold in liver within 14 days after the injection of triethyl Pb chloride.

Effects of Lead on Renal Heme Enzymes

Vij et al. (1998) explored Pb-induced alterations in male rats in the heme synthesizing enzymes, ALAD, and uroporphyrinogen I synthetase, and the effect of ascorbic acid supplementation in reversing these alterations. Lead-treated rats were injected IP with 20 mg/kg of Pb-acetate for 3 consecutive days and sacrificed 4 days later. A separate group of animals were administered 100 mg/kg ascorbic acid PO for 3 days following Pb administration. Blood Pb concentration was 4.67 ± 1.49 μg/dL in control rats, 16.59 ± 4.65 μg/dL in Pb-treated rats, and 7.83 ± 2.03 μg/dL in the Pb + ascorbic acid treated rats. Lead content of liver and kidney followed the same pattern. Blood ALAD activity was diminished in the Pb-treated rats but was restored in the Pb + ascorbic acid-treated rats. Uroporphyrinogen I synthetase activity followed the same pattern in blood but was not restored by ascorbic acid in liver. Total and nonprotein

sulfhydryl concentrations in blood were depressed by Pb administration and were not restored by ascorbic acid. However, levels in liver and kidney were restored by ascorbic acid.

ALAD levels following administration of Pb were also investigated by Rodrigues et al. (1996) and Peixoto et al. (2004). The study by Rodrigues et al. (1996) examined rats from Pb-exposed mothers that were maintained after weaning on either 0.5 or 4.0 mM Pb-acetate in drinking water for 21 days or 6 months. At sacrifice, ALAD activity was measured in kidney, forebrain, and cerebellum. Both 6-month-old Pb-exposed groups showed an increase in the kidney-to-body weight ratio, suggesting Pb-induced cell proliferation in the kidney. Blood Pb increased from 6.53 to 7.61 µg/dL in the 21-day-old exposed rats compared to 6-month-old controls. In the 0.5 mM Pb-treated group, blood Pb was 9.77 µg/dL in the 21-day-old and 41.63 µg/dL in 6-month-old rats, while in the 4.0 mM group, blood Pb was 44.35 µg/dL in the 21-day-old and 116.91 µg/dL in the 6-month-old group. ALAD activity was reduced at 6 months in the forebrain of the 4.0 mM Pb-treated group, and in the kidneys at 6 months in both the 0.5 mM and 4.0 mM Pb-treated groups. The study by Peixoto et al. (2004) examined the in vitro sensitivity (IC_{50}) to Pb of ALAD activity of brain, kidneys, and liver from suckling rats aged between 1 and 5, 8 and 13, or 17 and 21 days. The metal concentrations ranged from 0 to 50 µM for Pb-acetate. Rats in the first age group showed the greatest sensitivity in all three organs. Liver was the least sensitive to ALAD inhibition by lead, while brain was the most sensitive.

Effects of Lead on NaK-ATPase

Fox et al. (1991) explored the effect of in vivo Pb exposure on adult rat retinal and kidney NaK-ATPase. Pups, exposed to Pb through the milk of dams consuming 0, 0.02, or 0.2% Pb solutions, had mean blood Pb concentrations of 1.2, 18.8, and 59.4 µg/dL at weaning, respectively, and 5 to 7 µg/dL as 90 to 100-day-old adults. Prior Pb exposure produced significant dose-dependent decreases in isolated retinal NaK-ATPase activity (−11%; −26%), whereas activity in the kidney was unchanged. In contrast, NaK-ATPase from both isolated control tissues was inhibited by Pb in vitro. The half-maximal inhibitory dose of Pb for retinal and renal NaK-ATPase was 5.21×10^{-7} and 1.25×10^{-5} M, respectively. Retinal and renal NaK-ATPase were 20-fold and 1.1-fold more sensitive to inhibition by Pb than calcium. The

increased sensitivity of retinal, compared to renal, NaK-ATPase to inhibition following in vivo or in vitro Pb exposure may be related to their different α subunit composition.

Kramer et al. (1986) had also explored the half-maximal inhibitory dose for Pb-chloride on renal cortical homogenate NaK-ATPase, and found it to be 7×10^{-5} M. There was a competitive inhibition with regard to the substrate, ATP. Of several metals tested, Pb was second only to Hg in potency as a NaK-ATPase inhibitor.

Weiler et al. (1990) studied the effect of Pb on the kinetics of purified (from hog cerebral cortex) NaK-ATPase and potassium-stimulated p-nitrophenylphosphatase (K-pNPPase), which is referred to as the E2 configuration of the NaK-ATPase system. IC_{50} for Pb was found to be 8.0×10^{-5} M for NaK-ATPase and 5.0×10^{-6} M for K-pNPPase. Inhibition of NaK-ATPase by Pb was found to be noncompetitive with respect to K, but competitive with respect to Na and MgATP. Inhibition of K-pNPPase by Pb was competitive with respect to K.

Effects of Lead on Cardiovascular Hormones

Effects of Lead on Endothelin

Khalil-Manesh et al. (1993a) examined the role of endothelial factors in Pb-induced hypertension. They found that low Pb administration (0.01%), but not high Pb administration, (0.5%) resulted in increased blood pressure in rats treated for 12 months. In the low-Pb-treated rats, measurement of plasma endothelins-1 and -3 revealed that endothelin-3 concentration increased significantly after both 3 months (lead, 92.1 ± 9.7 vs. control, 46.7 ± 12.0 pmol/ml; $p < 0.001$) and 12 months (lead, 105.0 ± 9.3 vs. control, 94.1 ± 5.0 pmol/ml; $p < 0.01$), while endothelin-1 was unaffected. Plasma and urinary cyclic GMP concentrations, as a reflection of endothelium-derived relaxing factor (EDRF), decreased significantly at 3 months (plasma lead, 1.8 ± 0.9 vs. control, 4.2 ± 1.6 pmol/ml; $p < 0.001$) and 12 months (plasma Pb 2.2 ± 0.7 vs.control, 4.2 ± 0.9 pmol/ml; $p < 0.001$). High levels of Pb exposure did not result in hypertension, perhaps related to the fact that plasma concentrations of endothelin-1, endothelin-3, and cyclic GMP were unaltered at 3 months, while their concentrations were significantly decreased at 12 months (plasma cyclic GMP at 12 months, 2.2 ± 0.7, lead, vs. 4.2 ± 0.9 pmol/ml, control; $p < 0.001$). Thus, the path to development of hypertension in low-Pb rats was thought to be through an increase in the concentration of the vasoconstrictor, endothelin-3, and a decrease in the vasodilator hormone, endothelium-derived relaxing factor or NO.

Novak and Banks (1995) studied the effects of Pb on the actions of endothelin. They measured renal clearances and mean arterial pressure in rats in which endothelin-1 was infused at 110 ng/kg/min for 30 min. Lead was infused as Pb-acetate throughout the experiment at 0.48, 4.8, and 24 nmoles/min. At the two higher doses, Pb significantly attenuated the endothelin-induced increase in mean arterial pressure; Pb infused as 0.48 nmoles/min had no effect. An endothelin-induced decrease in GFR in control rats was completely blocked at the higher doses of lead. In additional experiments, calcium chloride was infused at 500 nmoles/min for 105 min, and then calcium + Pb (4.8 nmoles/min) were infused for another 105 min. In these experiments, there was no Pb-induced inhibition of the mean arterial pressure response to endothelin. However, the GFR response to the peptide remained blocked. These data illustrate that Pb inhibits the cardiorenal actions of endothelin and that a calcium-related process is involved in the systemic, but not the renal, component of this inhibition.

Effects of Lead on the Catecholamine System

Carmignani et al. (2000) studied the effects of low Pb exposure (60 ppm of Pb-acetate), given for 10 months, on catecholamine and monoaminoxidase (MAO) levels. Plasma catecholamines were measured by HPLC and MAO in aorta, liver, heart, kidney, and brain by a histochemical technique. Plasma norepinephrine (NE) was increased by 104% and adrenaline by 81% with no changes noted in L-DOPA and dopamine levels. MAO activity was increased in all organs. These workers ascribed the low Pb-induced hypertension in part to raised catecholamines levels.

Tsao et al. (2000) and Chang et al. (2005) measured changes in the β-adrenergic system in Wistar rats during and following Pb exposure. In Tsao et al. (2000), rats were chronically fed with 0.01, 0.05, 0.1, 0.5, 1.0, and 2.0% Pb-acetate for 2 months. Plasma catecholamine levels were measured by HPLC; cAMP levels in heart, kidney, and aorta by radioimmunoassay; and β-adrenergic receptors in heart, kidney, and aorta membranes by a radio ligand binding assay. Blood Pb increased from 0.05 ± 0.05 μg/dL in controls to 85.8 ± 4.1 μg/dL in the 2.0% Pb-treated group. Plasma NE, but not E, levels increased with increasing Pb dosage. β-Adrenoreceptor density of heart and kidney decreased progressively with increasing Pb dosage, whereas kidney β-adrenoreceptor density increased up to the 0.5% Pb group and then remained constant. Unstimulated cAMP was constant in all tissues, but cAMP stimulated by

isoprotorenol was lowered progressively in aorta and heart and increased in kidney. Chang et al. (2005) continued these measurements in rats fed 2% Pb-acetate for 2 months then withdrawn from Pb for periods of 1, 2, 3, 4, 5, 6, and 7 months. Blood Pb levels, systolic and diastolic blood pressure levels, and plasma NE were reduced after cessation of Pb exposure. This occurred in conjunction with an increase in β-adrenoreceptor density in heart and aorta and a decrease in β-adrenoreceptor density in kidney. (See Table AX5-5.5 for experimental details on these studies).

Effects of Chelators (Single or Combined) on Lead Mobilization

Effects of DMSA Alone

Cory-Slechta (1988) studied the mobilization of Pb by DMSA, following a 3- to 4-month exposure to 50 ppm of Pb-acetate in rats. These rats received an IP injection of saline or 25 or 50 mg/kg of DMSA once daily for either 1, 2, 3, 4, or 5 days. Tissue analyses indicated that DMSA mobilized Pb from blood, brain, kidney, and liver with no loss noted from femur.

Pappas et al. (1995) reported on Sprague-Dawley rats exposed to 550 or 1100 ppm Pb-acetate for 35 days and treated either with Pb + DMSA or DMSA alone at varying dosage for 21 days. Animals showed a dose-related reduction in Pb content of blood, brain, femur, kidney, and liver whether they received DMSA alone or Pb + DMSA.

Smith and Flegal (1992) studied the influence of DMSA on the mobilization and redistribution of Pb in skeletal and soft tissue compartments of low-Pb-exposed female rats, using stable Pb isotope tracer techniques. Rats reared on a low-Pb diet received ^{206}Pb-enriched drinking water for 1.5 days and then were chelated with a single IP injection of 0.11 mmol/kg dose of DMSA. Blood, kidney, brain, tibia, urine, and feces were collected 24 h after chelation and analyzed for Pb concentrations and for Pb isotope compositions. DMSA chelation significantly increased the diuresis of labile soft tissue Pb but not skeletal Pb. DMSA also appeared to cause the redistribution and input of a comparable amount of Pb to the skeleton and smaller relative amounts of Pb to the soft tissues of the chelated animals.

Varnai et al. (2001) determined whether ongoing Pb exposure influenced the mobilization of Pb in suckling rats. Six-day-old Wistar rats were given Pb-acetate in a dose of 2 mg/kg/day for 8 consecutive days. A treated group received a daily dose of 0.5 mmol/kg of DMSA PO six times on days 1 to 3 and 6 to 8. DMSA efficiently reduced Pb concentration in carcass, liver,

kidneys, and brain by approximately 50% versus with untreated controls. The results indicate that DMSA is an efficient oral chelator, even when challenged with ongoing Pb exposure.

Effects of Combined Chelators

Flora et al. (1994) compared the combined use of $CaNa_2EDTA$ with DMSA on the distribution of Pb and Pb-related biochemical effects with the influence of each chelator used alone. Wistar rats were given 1000 ppm Pb as Pb-acetate in drinking water for 4 months. They were then treated for 5 days with either saline, DMSA, 25 mg/kg PO twice daily; $CaNa_2EDTA$, 75 mg/kg once daily; or DMSA, 25 mg/kg twice daily, all followed by a single daily IP injection of 75 mg/kg of $CaNa_2EDTA$. Blood ALAD was reduced from 6.54 ± 0.18 nmol/min/ml in controls to 0.84 ± 0.10 in Pb-treated animals, with restoration to 3.03 ± 0.29 after combined treatment. Lead content in blood, liver, kidney, brain, and femur followed the same pattern: controls had 2.11 ± 0.23 µg/dL; Pb-treated, 46.0 ± 4.1 µg/dL; combined chelator-treated, 12.8 ± 0.3 µg/dL. Treatment with either DMSA or $CaNa_2EDTA$ alone produced intermediate results. Tandon et al. (1994) reported similar results.

Jones et al. (1994) compared the effects of DMSA, $CaNa_2EDTA$, $ZnNa_2EDTA$, and $ZnNa_3DTPA$ on Pb mobilization in mice. Mice were given 10 IP injections of Pb-acetate, 5.0 mg/kg per injection. Three days after the final Pb injection, mice received one of the chelators. Injections were given at a dose of 1 mmol/kg/day IP for either 4 days or 8 days. At 8 days, DMSA was the most effective chelator in removing Pb from kidney and bone. $CaNa_2EDTA$ was more effective in removing brain lead. When animals were loaded with 100 mg of Pb per kg body weight, DMSA remained more effective in removing Pb from kidney and bone while $CaNa_2EDTA$ was more effective in brain.

Kostial et al. (1999) evaluated the efficacy of three chelating agents, administered either as monotherapy or as combined treatments, in suckling rats. Lead acetate (5 mg Pb/kg IP) was administered to 7-day-old rat pups on experimental day 1, and chelating agents was administered on experimental days 2 and 3. The pups were divided into untreated control, EDTA-treated, meso-DMSA-treated, racemic DMSA-treated, EDTA plus meso-DMSA-treated, and EDTA + plus racemic DMSA-treated. Rats were killed on experimental day 5 and tissue analyses were done for lead, zinc, and copper. Treatment with EDTA did not affect tissue lead, but it reduced zinc in the carcass and liver. Meso-DMSA reduced Pb in the kidneys and brain and did not

affect organ essential elements. Racemic DMSA most efficiently reduced Pb concentrations in the carcass, kidneys, and brain, but it also reduced zinc and copper in the liver and zinc in the kidneys. Combined treatments with EDTA did not improve the efficiency of either DMSA isoform in decreasing tissue lead, but they did reduce tissue zinc concentrations. The results suggest that meso-DMSA may be the treatment of choice in acute Pb poisoning in infants, reducing Pb without affecting trace elements.

Malvezzi et al. (2001) evaluated the effects of DMSA, L-arginine (a precursor of NO), and the association of L-arginine and DMSA on tissue Pb mobilization and blood pressure levels in Pb-intoxicated rats. Tissue Pb levels and blood pressure evolution were evaluated in rats exposed to Pb (750 ppm in drinking water for 70 days), Pb + water for 30 more days , Pb + DMSA (50 mg/kg day, PO), L-arginine (0.6% in drinking water), the combination of L-arginine + DMSA for 30 more days, and their respective matching controls. Lead exposure increased Pb levels in the blood, liver, femur, kidney, and aorta. Lead levels in tissue decreased after cessation of Pb administration, except in the aorta. Blood Pb decreased from 67.8 µg/dL to 11.2 µg/dL in those subsequently treated with water, to 13.8 µg/dL in animals treated with Pb + DMSA, to 11.6 µg/dL in animals treated with Pb + L-arginine, and to 6.1 µg/dL in animals treated with Pb + L-arginine + DMSA. Lead mobilization from the aorta was only effective with the L-arginine/DMSA treatment. Lead administration increased blood pressure starting from the week 5, while L-arginine and DMSA treatments and the combination of L-arginine + DMSA decreased blood pressure levels of intoxicated rats; but these levels did not reach those of nonintoxicated rats. Treatment with L-arginine + DMSA was more effective than individual treatments in mobilizing Pb from tissues and in reducing the blood pressure of intoxicated rats. This paper lacks measurements of NO, which would have allowed the reader to more properly judge the mechanism of the effects of L-arginine administration. Furthermore, the dose of Pb was higher than in earlier studies that showed that DMSA was effective in lowering blood pressure. These and other studies are summarized in Tables AX5-7.3 and AX5-7.4.

Effects of Other Metals on Lead Distribution

Lead and Calcium

Fullmer (1992) published a review of intestinal interactions of Pb and calcium. High affinity Pb binding to intracellular calcium receptors and transport proteins, as well as the

involvement of Pb in calcium-activated and calcium-regulating processes, are thought to provide a partial molecular basis for the cellular and systemic effects of lead.

Maldonado-Vega et al. (1996) examined the intestinal absorption of Pb and bone mobilization during lactation. All experiments were started with 3-week-old female Wistar rats. Rats were impregnated at 16 weeks and were fed a 100 ppm solution of Pb-acetate for 158 or 144 days (mid-lactation or before lactation). Rats were also exposed for only 14 days, from 144 to 158 days (i.e., only during lactation). Nonpregnant rats from the same litter were exposed to Pb for periods equivalent to each of these groups. In the nonpregnant rats, blood Pb increased to 27.3 µg/dL from 5.2 µg/dL in controls. Similarly, kidney Pb increased to 13.2 nmol/g from 0.5 nmol/g, and bone Pb increased to 88.9 nmol/g from 0.9 nmol/g. ALAD activity decreased to 410 nmol/h/ml from 1004 nmol/h/ml. Compared to nonpregnant rats, there was a moderate increase in blood Pb in the lactating animals whether the Pb was given to mid-lactation or up to the period before lactation. Similarly, when Pb was administered only during lactation, there was a much higher increase in blood Pb in the pregnant rats than in the nonpregnant rats. Bone Pb concentration increased when Pb was given only during lactation, whereas bone Pb decreased (compared to Pb-treated nonpregnant rats) when the Pb was given either before lactation or before and during lactation. The authors considered that resorption of Pb from bone was the main additional source of Pb during lactation. The data indicate that Pb stored in bone as a result of prior maternal exposure should be considered as a major source of self intoxication and of Pb in milk available to suckling pups.

Lead and Cadmium

Skoczynska et al. (1994) compared the effects of the combined exposure to Pb and cadmium to each metal singly on tissue composition of trace metals. Experiments were performed on 5- to 6-week-old male Buffalo rats given Pb-acetate (70 mg lead/kg body weight twice a week) and cadmium chlorate (20 mg Cd/kg body weight once a week) intragastrically for 7 weeks either singly or in combination. Blood Pb in the control group was 5.1 µg/dL, compared to 29.6 µg/dL in the Pb-treated group. In contrast, the Pb + cadmium group showed a blood Pb of 37.4 µg/dL. After combined exposure to Pb and cadmium, the level of these metals in the liver and kidney was lower than after the single administration of Pb or cadmium. Exposure of the rats to cadmium resulted in an increase of kidney zinc and copper and liver zinc

concentrations; combined exposure to Pb + cadmium did not produce more extensive changes in tissue zinc and copper concentrations.

Lead and Selenium

Othman and Missiry (1998) examined the effect of selenium against Pb toxicity in male rats. Male albino rats were given a single dose of Pb-acetate (100 µmol/kg body weight) and sacrificed 3 or 24 h later. Another group of animals was pretreated with sodium selenite (10 µmol/kg body weight) 2 h before receiving Pb-acetate and sacrificed 24 h later. Selenium is well known as an antioxidant and cofactor for GSHPx. In this experiment, GSH content, GSHPx, SOD activities, and the products of lipid peroxidation (i.e., TBARS) were determined. It was found that lipid peroxidation was prevented and the reduction in GSH caused by Pb in liver and kidney was diminished by selenium. Lead-induced diminution in SOD activity and GSHPx activity was also returned to normal by selenium.

Tandon et al. (1992) studied the effect of selenium supplementation during chelation of Pb with $CaNa_2EDTA$. Rats were given Pb-acetate 10 mg/kg/day by gastric gavage for 6 weeks. This was followed by a 5-day treatment course of $CaNa_2EDTA$, 0.3 mmol/kg IP or of $CaNa_2EDTA$ + sodium selenite, 0.5 mg/kg PO. Selenium had marginal effects on Pb removal by $CaNa_2EDTA$ in blood, liver, and kidney and similar effects on ALAD activity.

Lead and Zinc

Flora et al. (1989) examined the role of thiamine, zinc, or their combination in the prevention or therapy of Pb intoxication. Albino rats received the following treatments daily through gastric gavage for 6 days each week over a six-week period, 10 mg/kg of Pb as Pb-acetate; or the same dose of Pb-acetate + thiamine (25 mg/kg) zinc sulfate (25 mg/kg) or Pb + thiamine and zinc. Rats that had been exposed to Pb only were additionally divided into four groups treated by gastric gavage daily for 6 days as follows: group I, water only; group II, thiamine only; group III, zinc only; and group IV, combined zinc + thiamine. The activities of blood ALAD, blood ZPP, blood lead, and urine ALA were determined. Blood Pb concentrations increased from 6.2 to 120.9 µg/dL, contrasting normal controls with Pb-treated animals. There was a slight reduction in blood Pb in animals treated with either thiamine or zinc and a greater reduction in animals treated with thiamine + zinc. In the post-Pb-exposure treatment group,

thiamine + zinc was also the most effective treatment. Liver and kidney Pb levels followed the same course but brain Pb was not reduced by treatment. Blood ALAD activity was decreased from a normal level of 7.63 µmol ALA/min/L to 0.69 in Pb-treated animals and restored to 7.52 in Pb + thiamine + zinc-treated rats. ZPP was increased from 1.78 µg/g hemoglobin to 4.22 in Pb-treated animals and reduced to 2.50 in Pb + thiamine + zinc-treated animals. Urine ALA was increased from 0.07 to 0.24 mg/dL in Pb-treated animals and decreased to 0.17 in Pb + thiamine + zinc-treated rats. Prevention was more effective than post-Pb-exposure treatment. This was thought to be due mainly to the decrease in the absorption of Pb in the GI tract in the presence of thiamine and/or zinc.

Flora et al. (1994) explored the dose-dependent effects of zinc supplementation during chelation of Pb in rats. The chelator employed was $CaNa_2EDTA$, whose toxic effects are known to be mainly due to the depletion of endogenous zinc and, possibly, copper and manganese. In this experiment, male Wistar rats were started on exposure to Pb-acetate, 10 mg/kg, administered through gastric gavage once daily for 56 days. Twenty-four hours later, the Pb-exposed animals were treated daily for 5 days as indicated: group I, saline ; group II, $CaNa_2EDTA$ 0.3 mmol/kg, IP, once daily for 5 days; group III, $CaNa_2EDTA$ + zinc sulfate, 10 mg/kg, PO once daily for 5 days; and group IV, $CaNa_2EDTA$ + zinc sulfate, 50 mg/kg, PO once daily for 5 days. Blood ALAD decreased from 6.30 to 1.44 nmol/min/mL erythrocyte in Pb-exposed animals, with no change after $CaNa_2EDTA$ treatment and partial restoration after the $CaNa_2EDTA$ + zinc, 10 mg/kg treatment. There was no improvement following zinc, 50 mg/kg. Lead concentration in blood increased from 4.6 µg/dL to 43.0 µg/dL in Pb exposed animals, decreasing to 22.5 µg/dL in $CaNa_2EDTA$-treated animals and decreasing further to 16.5 µg/dL in $CaNa_2EDTA$ plus zinc-treated animals. Zinc at 50 mg/kg led to an increase in blood Pb to 56.1 µg/dL. Changes in the liver follow the same pattern, while in the kidney, zinc increased the Pb levels further, and in the femur, zinc had no influence on Pb content. Blood zinc decreased from 6.1 to 5.7 µg/ml in Pb-exposed rats and further to 5.0 µg/ml in $CaNa_2EDTA$-treated animals. There was an increase to levels of 6.6 µg/ml on the 10 mg/kg supplement of zinc and a further increase to 8.1 µg/ml on the 50 mg/kg zinc supplement.

Lead and Iron

Hashmi et al. (1989) examined the influence of dietary iron deficiency, Pb exposure, or the combination of the two on the accumulation of Pb in vital organs of rats. Animals fed an iron deficient diet for 2 weeks were also subjected to orbital plexus puncturing twice a week to allow a Hb levels to decrease to 7 to 8 g/dL. Animals were thereafter treated for the next 6 weeks with iron deficient diets or iron-deficient diets + 0.1% Pb-acetate in drinking water. At the end of 3 and 6 weeks, animals from each group were sacrificed. Feeding of an iron-deficient diet during Pb exposure enhanced the accumulation of Pb in soft tissues and flat bones. For example, liver Pb content was 0.75 µg/g in control animals, 8.43 in Pb treated animals, and 12.93 in iron-deficient and Pb-treated animals. The sequence of events was similar in kidney, spleen, and femur except that the Pb content in femur was reduced in the iron deficient and Pb-treated group.

Singh et al. (1991) conducted a study to ascertain the role of iron deficiency during pregnancy in inducing fetal nephrotoxicity in mothers exposed to lead. Rats were fed either a normal iron diet or an iron free synthetic diet for 15 days, followed by a diet containing half of the daily required iron (47 mg/100 g ferrous ammonium sulfate) for a further 15 days. Female animals were mated with healthy adult males. Lead doses of 250, 500, 1000, and 2000 ppm were given in drinking water during pregnancy and lactation. Fetuses were removed by Caesarean section on the 21st day. Maternal blood Pb levels in rats on an iron deficient diet were higher than those in rats on a normal iron diet at all levels of Pb dosing. Similarly, placental Pb levels were higher in animals on an iron-deficient diet as compared to a normal diet. Lead content in the fetuses were higher on the iron-deficient diet. Lead administration resulted in dose-dependent hydropic degeneration of renal proximal tubular cells in the fetuses. At a dose of 2000 ppm Pb with iron deficiency, more Pb accumulated in maternal blood, placenta, and fetuses and maximum pathological changes were seen in the fetal kidney as compared to other doses.

Lead and Aluminum

Shakoor et al. (2000) reported beneficial effects of aluminum on the progression of Pb-induced nephropathy in rats. Male albino rats were treated with water only or Pb-acetate (125 mg/kg) and/or aluminum chloride (50 mg/kg or 100 mg/kg) for a period of 90 days. Aluminum was found to prevent the Pb-induced increase in relative kidney weight in a dose-

dependent manner. Aluminum also prevented Pb-induced increases in plasma creatinine levels of Pb- treated animals. The net deposition of Pb in kidneys was lower in animals that were given both Pb-acetate and aluminum chloride simultaneously. By day 90, plasma creatinine was 1.26 mg/dL in control animals, 1.88 mg/dL in Pb-treated animals, and 1.34 and 1.44 mg/dL in Pb and aluminum-treated animals. Similarly, kidney Pb increased from 5.4 µg/g in control animals to 220.0 µg/g in Pb-treated animals and decreased to 138.5 and 98.9 µg/g in Pb and aluminum treated animals. These and other studies are summarized in Table AX5-7-5.

5.7.4.4 Effect of Age on Lead Toxicity

Han et al. (1997) examined the hypothesis that the high rate of bone remodeling during childhood and the consequent high calcium and Pb turnover would result in a substantial reduction in bone Pb stores, so that much of the Pb incorporated in bone during childhood does not persist into adulthood. They treated female Sprague-Dawley rats with 250 ppm of Pb in drinking water for 5 weeks beginning at 5, 10, or 15 weeks of age. Organ harvesting occurred 4 weeks after the end of Pb exposure for all groups, as well as 8 and 20 weeks after cessation of Pb ingestion in the rats exposed beginning at 5 weeks of age. Organs examined were brain, kidney, liver, femur, and spinal column bone. Blood and organ Pb concentrations were significantly higher in the rats exposed beginning at 5 weeks of age than in those exposed beginning at 10 or 15 weeks of age. The results of this experiment rejected the hypothesis and suggested instead that a younger age at Pb exposure is associated with greater Pb retention and toxicity, even in the absence of continued Pb exposure.

Garcia and Corredor (2004) examined biochemical changes in the kidneys after perinatal intoxication with Pb and/or cadmium. Lead acetate (300 ppm) and/or cadmium acetate (10 ppm) were administered in drinking water to pregnant Wistar rats from day 1 of pregnancy to parturition (day 0) or until weaning (day 21). The following kidney enzyme activities were determined: alkaline and acid phosphatases, Mg-ATPase, and NaK-ATPase. Blood Pb was measured in control pups as well as in pups exposed to lead at parturition and at weaning. Control pups showed 1.43 µg/dL of blood Pb compared to 31.5 µg/dL at day 0 and 22.8 µg/dL at day 21 in pups exposed to lead. In those rats receiving both cadmium and Pb, the blood Pb concentration was 23.2 µg/dL at day 0 and 13.2 µg/dL at day 21. Lead caused a significant inhibition of kidney alkaline phosphatase and kidney acid phosphatase. At parturition, Pb

intoxication produced a strong inhibition of NaK-ATPase (~80%) as well as of Mg-ATPase activities (~24%); whereas, when Pb was given in combination with cadmium, these inhibitory effects were attenuated. At weaning, Pb continued to produce a significant inhibition of Mg-ATPase but had no effect on NaK-ATPase. Thus, simultaneous perinatal administration of both Pb and cadmium seemed to protect against the toxicity produced by Pb separately.

Cory-Slechta (1990a,b) published two articles on the effects of old age on the disposition of lead. In the first study (1990a) male F344 rats, at the ages of 8 months (adult) and 16 months (old) were exposed to concentrations of 0, 250, or 500 ppm Pb-acetate in drinking water for 7 months. At these Pb doses, prior studies had indicated that blood Pb levels ranged from 60 to 90 µg/dL. Blood lead, ZPP, and urinary ALA levels were determined after both 3 and 7 months of exposure. Organ weights, tissue Pb concentrations, and urinary excretion of lead, calcium, copper, and zinc were examined after 7 months of exposure. Tissue Pb distribution was markedly altered in old rats: in bone and kidney, Pb levels were reduced while liver Pb was substantially increased. Blood Pb levels in adult and old rats were comparable at both measurement intervals, as was urinary Pb excretion at 7 months. Lead-induced elevation of ZPP exhibited differential changes between 3 and 7 months; values in adults declined while levels in old rats increased or remained unchanged. In the adult group, Pb exposure increased calcium excretion primarily at the 500 ppm exposure level. In contrast, Pb exposure decreased urinary calcium excretion in old animals at the higher exposure level. No effects of either age or Pb exposure were detected in the comparison of adult versus old urinary excretion of zinc or copper.

In the second study, Cory-Slechta (1990b), young (21 days old), adult (8 months old), and (16 months old) rats exposed to 0, 2, or 10 mg of Pb-acetate/kg per day for a period of 9.5 months were evaluated. Differences in the tissue distribution of Pb with age included lower bone levels, but increased concentrations in brain, liver, and kidney. Differences in blood Pb levels over the course of exposure were not remarkable. Thus, these effects did not appear to reflect an enhanced Pb absorption from the GI tract with age. Instead, the bone changes may reflect enhanced bone resorption with a concurrent decline in bone apposition with age, combined with altered patterns of urinary Pb excretion over time, i.e., elevated urinary Pb at 3 and 6 months, but comparable Pb excretion at 9.5 months, as compared to young and adult rats.

5.7.5 Summary

Highlights of the previous 1986 Pb AQCD and of studies done between 1986 and the present are outlined in this section.

1986 Document

- In animal studies, nuclear inclusion bodies were found in proximal tubules, identified as 27 kDa or 32 kDa proteins in combination with lead. Subsequently, a 63 kDa Pb-binding cytosolic protein was described in kidney.

- Swollen mitochondria, with diminished mitochondrial function, were found in the proximal tubules.

- Renal ALAD was the same in Pb-treated animals as in controls when GSH was present, but was reduced when GSH was absent.

Newer studies

- Hyperfiltration, when compared to age- and sex-matched normal controls, was found in adults who had suffered from childhood Pb poisoning, in young occupationally exposed Pb workers in Korea, and in both low-Pb-treated rats and high-Pb-treated rats up to 3 months of exposure. This is paralleled in animal experiments by an increase in kidney weight.

- Various new urinary markers for Pb toxicity have been described. These include NAG, β2-microglobulin, α1-microglobulin, retinol binding protein, GST, lysozyme, γ-glutamyl transferase, alanine aminopeptidase, prostanoids, and brush border antigens. The literature on these markers is voluminous, but, on review, only GST and α1-microglobulin seemed to be appropriate urinary markers. NAG, which has been most extensively investigated, appears in detailed-animal studies to be overly sensitive, increasing in low-Pb-treated animals, despite an absence of pathological changes on ultrastructural study. β2-Microglobulin, and possibly retinol binding protein, which are low-molecular weight proteins reabsorbed by the proximal tubule, appeared to be elevated only with high levels of blood Pb (>80 μg/dL).

- Animal studies have implicated free radicals in the pathogenesis of Pb-induced hypertension and renal disease. A sequence of free radicals can be demonstrated in Pb-induced disease, as evidenced by an increase in superoxide radicals, hydroxyl radicals, hydrogen peroxide, and peroxynitrite, together with a diminution in GSH in liver, brain, and aorta. Nitric oxide is most commonly decreased (by free radicals) as is urinary cyclic GMP. Aortic guanylate cyclase is decreased. The enzyme responsible for an increase in the production of free radicals, NAD(P)H oxidase, is increased by Pb, whereas eNOS and iNOS, the enzymes involved in the production of nitric oxide, are also increased, attesting to the importance of free radical destruction of nitric oxide. Antioxidants reverse these changes and diminish blood pressure.

- Norepinephrine and epinephrine are increased by Pb administration, whereas β-adrenoreceptor density of heart and kidney are decreased. In a second study, norepinephrine, but not epinephrine, was increased by Pb.

- Various antioxidants have been used in conjunction with chelators, to both remove Pb from tissue and to diminish free radicals. Taurine, lipoic acid, arginine, ascorbic acid, vitamin E, thiamine, tempol, and lazaroids have been used in conjunction with DMSA, all improving free radical diminution.

- Metal combinations have also been employed to reduce tissue Pb and/or affect free radicals. Cadmium increases Pb in blood when both are given, but diminishes Pb in liver and kidney. Selenium, an antioxidant, improves both parameters, as does thiamine or L-lysine plus zinc. Iron deficiency increases intestinal absorption of Pb and the Pb content of soft tissues and bone. Aluminum decreases kidney Pb content and serum creatinine in Pb-intoxicated animals.

- Age also has an effect on Pb retention. There is higher Pb retention at a very young age and lower bone and kidney Pb at old age, attributed in part to increased bone resorption and decreased bone accretion.

5.8 EFFECTS ON BONE AND TEETH

5.8.1 Biology of Bone and Bone Cells

By weight, bone is composed of 28% collagen fibers (predominantly type I collagen) and 5% noncollagenous proteins (osteocalcin, osteonectin, and other proteoglycans), with crystals of hydroxyapatite [$Ca_{10}(PO_4)_6(OH)_2$] making up the remaining 67%. In addition to providing mechanical support for the body and protection of vital organs, the skeletal system also functions in a metabolic capacity. Historically, bones have been classified as either long or flat based on their appearance, with long bones including limb bones, e.g., the femur and humerus, and flat bones including the bones of the skull, sternum, pelvis, and scapula. Long and flat bones originate by distinct methods of formation, endochondral and intramembranous, respectively, with long bones eventually using both processes. In endochondral bone formation, a mineralized, cartilaginous matrix precedes the transition into true bone, while in intramembranous formation, the bone forming cells create bone directly without the cartilaginous template.

Bone cells responsible for producing the bone matrix of collagen and ground substance are called osteoblasts. Several signaling factors including growth factors and hormones

influence pre-osteoblastic cells to differentiate into mature osteoblasts and subsequently synthesize and mineralize the extracellular matrix to form mature bone. It is during the process of bone mineralization that the Pb ion (Pb^{2+}) can become incorporated by substituting for the calcium ion (Ca^{2+}). The bone cells responsible for bone resorption are the osteoclasts. Osteoclasts, which are large and multicellular (4 to 20 cells), dissolve bone matrix and hydroxyapatite by synthesizing and releasing lysosomal enzymes and acidifying the extracellular surroundings. It is during the process of dissolving bone, or demineralization, that Pb stored in bone can be released locally and into the general system.

Bone cell function may be compromised both directly and indirectly by exposure to Pb. Regulation of bone cells occurs by numerous local and systemic factors, including growth hormone (GH), epidermal growth factor (EGF), transforming growth factor-beta 1(TGF-β1), and parathyroid hormone-related protein (PTHrP). As discussed further below in this section, the presence of lead can potentially interfere with each of these factors. The bones of the skeleton serve as the primary reservoir for calcium and phosphate in the body and help to maintain homeostasis of these ions in the serum through bone turnover or remodeling. Vitamin D [1,25-$(OH_2)D_3$] maintains the normal range of calcium in the serum by increasing the efficiency of calcium absorption in the intestines and facilitating differentiation of stem cells into osteoclasts, which break down bone and mobilize calcium (and lead) stores. Parathyroid hormone (PTH), in turn, regulates the production of vitamin D in the kidney. Lead has been shown to interfere with the action of both of these hormones. Other substances influenced by lead and discussed in this section are alkaline phosphatase, an enzyme necessary for mineralization of bones and teeth, and osteocalcin, a noncollagenous protein whose spatial and temporal pattern of expression suggests a role in bone mineralization. Both substances are also markers for osteoblast activity and, by default, bone formation. Alkaline phosphatase is a potential carrier of ionic calcium and is capable of hydrolyzing inhibitors of mineral deposition such as pyrophosphates.

5.8.2 Summary of Information Presented in the 1986 Lead AQCD

Lead has been shown to become localized and accumulate in bones and teeth, with accumulation beginning as early as fetal development. Lead administered to rats as a single dose results in blood lead concentrations that are initially elevated, but rapidly fall as Pb is transferred

to bone or excreted. The dose of Pb administered does not apparently affect distribution to the various body compartments; however, the rate-limiting step in the clearance of Pb from rats and mice involves absorption into/clearance from the skeletal system. The loss of Pb from various organs and tissues follows first-order kinetics, except from bone. More absorbed Pb is retained by young animals compared with adult animals, leading to higher tissue levels. Moreover, once Pb is incorporated into the young animal's body, the long-term rate of retention is greater than that of adults. In Pb-exposed animals, Pb is distributed subcellularly, preferentially to the nucleus and mitochondrial fractions.

During lactation in mice, a redistribution of tissue Pb occurs (mobilization), resulting in the transfer of Pb and calcium from mother to pups via the milk, and subsequent overall loss of Pb in the mothers. Lead transfer to suckling rats via mother's milk has been reported to be approximately 3% of the maternal body burden or more, if Pb exposure continues during lactation. Eight days after a single injection of Pb, the content of Pb in rabbit's milk was 8-fold higher than the maternal blood level, suggesting Pb transfer can occur against a concentration gradient. Transplacental transfer of Pb from mother to fetus also occurs in various animals.

In rats, a significant reduction of calcium in the diet leads to enhanced uptake of lead into the bones and other tissues. In general, an enhanced uptake of Pb into tissues is also seen in rats fed diets deficient in iron, zinc, copper, or phosphorus, and in the presence of low or excess vitamin D.

5.8.3 Bone Growth in Lead-Exposed Animals

Lead is readily taken up and stored in the bone of experimental animals, where it can potentially manifest toxic effects that result in stunted skeletal growth. In experiments reported since the 1986 Pb AQCD, Hac and Krechniak (1996) determined uptake and retention of Pb in bone from rats exposed to plain water or water containing Pb-acetate (41.7 to 166.6 mg/L) for 12 to 16 weeks. After 4 weeks, the skeletal Pb in animals receiving the lowest dose was almost 5 times higher than control animals (5.9 versus 1.2 µg Pb/g bone, respectively). Lead levels in bones from animals receiving 83.3 mg/L and 166.6 mg/L were dose-dependently higher at 11.7 and 17.0 µg Pb/g bone, respectively, after 4 weeks of exposure. All bone Pb levels were maintained essentially in a steady state until the completion of exposure, when all animals were placed on control water. Approximately 64% of Pb remained in the bones of rats in the

83.3 mg/L exposed group at 64 days postexposure. Similarly, airborne Pb can be inhaled and subsequently incorporated into bone. Grobler and co-workers (1991) exposed 6-week-old rats to either "clean air" (0.05 μg Pb/m^3) or air containing 77 μg Pb/m^3 and found significant differences in the amount of Pb incorporated into the alveolar bones of the animals. After 70 days, a mean of only 0.2 μg Pb/g of bone dry mass was found in bone from control animals, while 16.9 μg Pb/g was present in bone from the 77 μg Pb/m^3 exposure group. Exposure to air containing 249 μg Pb/m^3 for 28 days or 1,546 μg Pb/m^3 for 50 days, resulted in mean values of 15.9 and 158 μg Pb/g dry weight of Pb incorporation into the bone, respectively, highlighting the fact that dose and length of exposure are determinates of amount of Pb contained in the bones of these animals. The uptake of Pb by bone has the potential for immediate toxic effects on the cellular processes occurring during bone growth, development, and maintenance, with the additional potential for delayed toxicity from release of stored Pb during periods of normal or accelerated bone remodeling.

Numerous studies have examined growth suppression associated with developmental Pb exposure. Hamilton and O'Flaherty (1994) examined the effects of Pb on growth in female rats, and subsequently, on growth and skeletal development in their offspring. Administration of drinking water containing either 250 or 1,000 ppm lead to weaning female rats for 49 days produced no alteration in growth rate in these future dams. The rats were then bred, with Pb exposure continuing through parturition and lactation. Lead did not affect gestation time nor Day 1 suckling body weight, however, pup body weight and tail length were subsequently decreased in both exposure groups. A 10% increase in tibial growth plate width and disruption of chondrocyte organization were observed in offspring from the high exposure group.

In male rats exposed to 100 ppm Pb in drinking water and a low calcium diet for up to one year, bone density was significantly decreased after 12 months, while rats exposed to 5,000 ppm Pb had significantly decreased bone density after 3 months (Gruber et al., 1997). Pb content of femurs was significantly elevated over the content of control rats at all time points (1, 3, 6, 9, 12 months). Trabecular bone from the low dose animals was significantly decreased from 3 months forward. Young female rats exposed to 17 mg of Pb-acetate per kg of feed for 50 days showed no differences in the length of the femurs, but the mean length of the 5th lumbar vertebra was significantly decreased (Gonzalez-Riola et al., 1997; Escribano et al., 1997). The mean length of the femur growth plate cartilage was also significantly decreased in Pb-exposed animals.

In a dose-response study, Ronis et al. (1998a, 1998b) exposed pregnant rats to Pb-acetate in drinking water (0.05% up to 0.45% w/v) beginning at gestation Day 5 and continuing through weaning of offspring at Day 21. Early bone growth was significantly depressed in a dose-dependent fashion in pups of all Pb-exposed groups, with growth suppression in male offspring considerably greater than in females. Significant decreases in plasma insulin-like growth factor and plasma sex steroids and increased pituitary growth hormone were also observed. This is somewhat in contrast to the findings of Camoratto and coworkers (1993), who reported low exposure to 0.2% Pb nitrate (125 ppm Pb) did not significantly affect growth, though males weighed significantly less than females. Between age 57 and 85 days Ronis et al. (1998b) noted that growth rates were similar in control and Pb-exposed pups, suggesting exposure at critical growth periods such as puberty and gender may account for differences in growth reported by various investigators. In a series of follow-up experiments (Ronis et al., 2001) reported a dose-dependent decrease in load to failure in tibia from Pb-exposed (0.15% and 0.45% Pb-acetate in drinking water) male pups only. Hormone treatments (estradiol in females or L-dopa, testosterone or dihydrotestosterone in males) failed to attenuate Pb deficits during the pubertal period. Distraction osteogenesis experiments performed after stabilization of endocrine parameters (at 100 days of age) found decreased new endosteal bone formation and gap x-ray density in the distraction gaps of Pb-exposed animals (Ronis et al., 2001).

Hamilton and O'Flaherty (1995) found Pb disrupted mineralization during growth when they implanted demineralized bone matrix subcutaneously into male rats. In the matrix that contained 200 μg Pb/g of plaque tissue, alkaline phosphatase activity and cartilage mineralization were absent, though calcium deposition was enhanced. Separate experiments found enhanced calcification and decreased alkaline phosphatase activity in rats implanted with a control (no Pb) matrix and given 1,000 ppm Pb in drinking water for 26 days.

In summary, results from animal studies suggest Pb exposure is capable of adversely affecting bone growth and density, potentially manifesting its action through interference with growth and hormonal factors as well as toxic effects directly on bone.

5.8.4 Regulation of Bone Cell Function in Animals – Systemic Effects of Lead

Lead may exhibit multiple complex systemic effects that ultimately could influence bone cell function. As discussed in the animal studies below, Pb can modulate alterations in calcium binding proteins and in calcium and phosphorus concentration in the blood stream, in addition to potentially altering bone cell differentiation and function by altering plasma levels of growth hormone and calciotropic hormones such as vitamin D_3 [1,25-$(OH_2)D_3$] and parathyroid hormone.

5.8.4.1 Hypercalcemia/Hyperphosphatemia

Intravenous injection of Pb has been shown to produce both an acute hypercalcemia and hyperphosphatemia in rats (Kato et al., 1977). Injection of a relatively high dose of 30 mg/kg Pb resulted in maximum values of calcium (17 mg%) after one hour and maximum values of phosphorus (13.5 mg%) after 30 minutes. After 12 hours the levels of both calcium and phosphorus had returned to baseline levels. Histochemical examination demonstrated deposition of Pb into bone and dentin in the rats, suggesting a direct action of Pb on bone and/or teeth, ultimately displacing calcium and phosphorus and thereby producing hypercalcemia and hyperphosphatemia.

5.8.4.2 Vitamin D [1,25-(OH2)D3]

As discussed above, vitamin D [1,25-$(OH_2)D_3$] modulates the normal range of calcium in serum. In rats fed a low calcium or low phosphorus diet, ingestion of 0.82% Pb in the diet reduced plasma levels of 1,25-$(OH_2)D_3$; however, this effect is lost when a high calcium or normal phosphorus diet is given (Smith et al., 1981), suggesting a high calcium/phosphorus diet reduces the susceptibility of vitamin D system to the effect of Pb. No mobilization of calcium from bone or elevation of inorganic phosphorus was seen. Ronis et al. (2001) also reported no effects of Pb on plasma concentrations of vitamin D metabolites, 25-OH D_3 or 1,25-$(OH_2)D_3$, in pubertal male rats exposed to either 0.15% or 0.45% Pb acetate in drinking water and maintained on an adequate diet. Fullmer (1995) found vitamin D function was severely compromised in young growing chicks given a diet low in calcium (0.1% calcium) for two weeks and then exposed to 0.2% or 0.8% Pb in their diet for an additional one or two weeks. In chicks

maintained on an adequate diet (1.2% calcium), exposure to 0.2% or 0.8% Pb in the diet resulted in increased plasma levels of 1,25-$(OH_2)D_3$ as well as significantly increased intestinal Calbindin-D protein [a calcium binding protein induced by 1,25-$(OH_2)D_3$] and its associated mRNA, when compared with unexposed control chicks. Levels of intestinal Calbindin-D mRNA and protein and plasma levels of 1,25-$(OH_2)D_3$ were elevated during the first week of Pb exposure to chicks fed a diet deficient in calcium, but were significantly decreased by the second week of Pb exposure. The study suggested Pb was mediating its effect through 1,25-$(OH_2)D_3$, rather than via a direct action on the Calbindin-D protein. Follow up studies by Fullmer et al. (1996) confirmed dose dependent increases in serum 1,25-$(OH_2)D_3$ levels (and Calbindin-D protein and mRNA) with increasing dietary Pb exposure (0.1% to 0.8%) in similar experiments performed on Leghorn cockerel chicks fed an adequate calcium diet.

5.8.4.3 Parathyroid Hormone

At least one animal study has associated experimental Pb exposure with secondary hyperparathyroidism. Szabo et al. (1991) exposed Wistar Kyoto rats to either 1% Pb acetate in water for a short term (10 weeks) or varying concentrations (0.001 to 1% Pb acetate) for a longer term (24 weeks) to assess the influence of Pb on the interaction of the parathyroids with 1,25-$(OH_2)D_3$. Short term administration of 1% Pb resulted in significant increases in bone Pb; however, total serum calcium and ionized serum calcium were significantly decreased, as compared to controls. Circulating levels of 1,25-$(OH_2)D_3$ were also decreased, though the rats were maintained on a normal calcium diet (0.95%). Parathyroid glands from rats exposed short term to Pb were significantly increased in size over those in control animals (178 µg per gland versus 96 µg per gland) and specific binding of 1,25-$(OH_2)D_3$ to parathyroid and intestinal tissue was increased. Likewise, long term administration of 1% Pb resulted in significant increases in bone Pb and normalized parathyroid gland weights, and a significant decrease in the level of 1,25-$(OH_2)D_3$. In the long term study, a dose-dependent increase in parathyroid weight occurred with increasing exposure to Pb in drinking water. The authors concluded the secondary hyperparathyroidism was associated with, and/or a result of, the hypocalcemia and decreased 1,25-$(OH_2)D_3$ levels secondary to Pb exposure.

5.8.4.4 Growth Hormone

As discussed in Section 5.8.3, exposure to Pb has been associated with altered bone metabolism and decreased growth and skeletal development (Hamilton and O'Flaherty, 1994, 1995; Gruber et al., 1997; Gonzalez-Riola et al., 1997; Escribano et al., 1997; Ronis et al., 1998a,b, 2001; Camoratto et al., 1993), suggesting perturbation of one or more endocrine factors such as growth hormone. To examine the effect of exposure to low-level Pb on pituitary growth hormone release, Camoratto et al. (1993) exposed pregnant female rats to 0.02% Pb nitrate (125 ppm Pb) beginning on gestational day 5 and continuing in pups through postnatal day 48. Basal release of growth hormone from control and Pb-exposed pups at age 49 days was not significantly different. Growth hormone releasing factor-stimulated release of growth hormone from pituitaries of Pb-exposed pups was smaller than the stimulated release of growth hormone from pituitaries of control animals (75% increase over baseline vs. 171% increase, respectively), but the difference did not achieve significance ($p = 0.08$). Growth hormone content of the pituitary glands was also not influenced by Pb exposure. Ronis et al. (1998b) reported similar findings in rat pups exposed to 0.05%, 0.15%, or 0.45% Pb acetate in drinking water from gestation day 5 through postnatal day 85, with the exception being significantly elevated pituitary growth hormone levels at postnatal day 55. Taken together, these rat studies suggest that differences in growth seen with Pb exposure may not necessarily be the result of alterations in secretion of growth hormone.

5.8.5 Bone Cell Cultures Utilized to Test the Effects of Lead

5.8.5.1 Bone Organ Culture

In an early bone organ culture study utilizing incorporated radioactive Pb into fetal radii and ulnae, Rosen and Wexler (1977) reported release of Pb as the concentration of calcium in the media was reduced or with addition of parathyroid hormone, but that calcitonin inhibited the release of Pb as expected, verifying the capacity of this model system. The bone organ system was subsequently used to evaluate the efficacy of Pb chelating agents, such as D-Penicillamine and $CaNa_2EDTA$ (Rosen and Markokwitz, 1980; Rosen et al., 1982).

5.8.5.2 Primary Cultures of Osteoclasts and Osteoblasts

The ability to isolate primary cultures of osteoclasts and osteoblasts from mouse calvaria provided an additional experimental model system to study the effects of Pb on specific bone cells. Using isolated osteoclasts and osteoblasts, Rosen (1983) reported that uptake of radioactive Pb by osteoclasts was rapid, almost linear, while osteoblasts showed very little increase in uptake of Pb at increasing media concentrations. Physiological concentrations of parathyroid hormone markedly increased uptake of Pb and calcium by osteoclast cells and, once loaded with Pb, osteoclasts were capable of releasing Pb slowly into the media. Further kinetic analysis of cultured osteoclastic bone cells indicated that cellular Pb is primarily associated with the mitochondrial fraction (~78%) and that this Pb is readily exchangeable with the outside media (Pounds and Rosen, 1986; Rosen and Pounds, 1988). Experiments conducted to characterize the steady-state kinetic distribution and metabolism of calcium and Pb supported the concept that the two elements are metabolized similarly in the osteoclasts cells (Rosen and Pounds, 1989).

5.8.5.3 Rat Osteosarcoma Cell Line (ROS 17/2.8)

In recent years, the rat osteosarcoma cell line ROS 17/2.8 has been used extensively to investigate the influence of Pb on various cellular processes and kinetics within these osteoblast-like cells. The ROS 17/2.8 cell model is useful in that the cells are capable of producing osteocalcin (a bone protein important for proper bone mineralization), have high alkaline phosphatase activity (an enzyme normally associated with mineralization of cartilage), possess vitamin D receptors, and respond to parathyroid hormone. In comparisons of cellular lead toxicity and metabolism between primary cell culture from mouse calvaria and the rat osteosarcoma cell line, Long and coworkers (1990a) reported remarkable similarities in the profile of radiolabeled Pb kinetics and intracellular Pb distribution. Using this cell line, Schanne and coworkers (1989) simultaneously measured intracellular Pb and calcium concentrations and found 5 and 25 micromolar Pb produced sustained 50% and 120% (respectively) increases in intracellular calcium over a 5 hour period, and that measurable entry of Pb into the cells could be demonstrated at the higher concentration. These findings advanced the hypothesis that perturbation of intracellular calcium concentration may be the mechanism of Pb bone toxicity. Schirrmacher and coworkers (1998) reported that calcium homeostasis is upset within

20 minutes of its addition to calvarial bone cell culture. Their results suggested that the calcium-ATPases of intracellular stores were potentially poisoned by Pb entering the cells. Wiemann et al. (1999) demonstrated that Pb was also capable of interfering with the calcium release activated calcium influx (CRAC) in calvarial bone cell cultures. Pb was found to partially inhibit the influx of calcium into the bone cells, plus influx of Pb into the cells was greatly enhanced (2.7 fold) after CRAC had been induced. These effects of Pb were found to be independent of any inhibitory effect on calcium-ATPase.

Miyahara et al. (1995) performed a series of experiments in ^{45}Ca-labeled bone organ culture to determine whether the Pb-induced hypercalcemia was the result of the active process of biological bone resorption or simply physiochemical mineral dissolution. Lead introduced into the culture at concentrations of 50 μM and above stimulated the release of calcium and hydroxyproline into the medium, however no release was elicited from bones inactivated by freezing and thawing. Pb-stimulated ^{45}Ca release was inhibited by eel calcitonin, bafilomycin A_1, and scopadulcic acid B, suggesting the release was secondary to osteoclastic bone resorption. Further evidence to support this conclusion came from experiments examining the influence of two inhibitors of cyclooxygenase on Pb-induced bone resorption. Lead was found to stimulate prostaglandin E_2 release and in cultures, there was a high correlation between prostaglandin E_2 released into the media and ^{45}Ca release. In the presence of cyclooxygenase inhibitors (blocking prostaglandin synthesis), Pb-stimulated ^{45}Ca release was inhibited suggesting the mechanism of bone resorption in this instance was via a prostaglandin E_2-mediated mechanism.

Lead has been demonstrated to directly impair production of osteocalcin by ROS 17/2.8 cells by 70% after 24 hours of exposure to 25 micromolar Pb (Long et al., 1990a). The resulting decrease in cell proliferation is in agreement with similar studies by Sauk et al., 1992). Interestingly, exposure of dental pulp cells, which also produce osteocalcin, to a similar concentration of Pb reduced osteocalcin production by 55% after 12 hours of exposure (Thaweboon et al., 2002). Vitamin D has been shown to increase osteocalcin production in ROS 17/2.8 cells; however, Pb inhibited the vitamin D-stimulated osteocalcin production in a dose-dependent manner from 0 up to 25 micromolar concentrations, plus was shown to be capable of attenuating basal (non-vitamin D-stimulated) osteocalcin production (Long et al., 1990a). Lead (5 to 20 micromolar) inhibition of vitamin D stimulation of osteocalcin in ROS cells was also reported by Guity and coworkers (2002). Later studies suggested that Pb acts by inhibiting

vitamin D activation of calcium channels and interferes with regulation of calcium metabolism (Schanne et al., 1992), though apparently this effect is not mediated via PKC (Guity et al., 2002). Angle and coworkers (1990) reported that 24 hours of incubation with vitamin D (10 nM) was capable of evoking a 4 to 5 fold increase in osteocalcin production and a 100% increase in cellular alkaline phosphatase activity in ROS cells. Osteocalcin production and cellular DNA contents were increased 100% and 20% respectively by addition of insulin-like growth factor (92.5 ng/mL). Consistent with a toxic effect of Pb on osteoblast function, the addition of 1 to 10 μM Pb to the system inhibited both basal and stimulated osteocalcin secretion, alkaline phosphatase activity and DNA contents (Angle et al., 1990). Dose- and time-dependent reduction in alkaline phosphatase activity with Pb exposure (2 to 200 micromolar) has also been reported in osteosarcoma cells, along with parallel reductions in steady state levels of alkaline phosphatase mRNA levels (Klein and Wiren, 1993). No effect on cell number or DNA and protein synthesis was seen at these levels of Pb exposure.

Though the exact mechanism of Pb toxicity on osteocalcin was unclear, Pb was known to inhibit some of the functional properties of osteocalcin including inhibition of osteocalcin adsorption to hydroxyapatite. An investigation by Dowd and coworkers (1994) utilized the ability of osteocalcin added to a solution of $^{43}CaCl_2$ to broaden ^{43}Ca resonance, as a method to examine binding of calcium to osteocalcin and the influence of Pb on calcium binding. It was determined that the dissociation constant of calcium for osteocalcin was 7 micromolar, while the dissociation constant for Pb was determined by competitive displacement to be 2 nM, indicating more than three orders of magnitude tighter binding of Pb than calcium to osteocalcin and the likelihood that even submicromolar levels of free Pb would significantly inactivate osteocalcin. Circular dichroism indicated that upon binding, Pb induces a similar structural change in osteocalcin to that found with calcium binding, but the binding with Pb occurs at 2 orders of magnitude lower than with calcium (Dowd et al., 2001). Similarly, hydroxyapatite binding assays indicated Pb causes an increased absorption to hydroxyapatite that is similar to calcium, but again at 2 to 3 orders of magnitude lower concentration, potentially leading to low bone formation rates and/or density (Dowd et al., 2001).

Besides perturbation of calcium metabolism, Pb has been shown to reduce intracellular free magnesium concentrations by 21% in osteosarcoma cells incubated in 10 micromolar Pb for 2 hours (Dowd et al., 1990). Under these same conditions, the unidirectional rate of ATP

synthesis (i.e. P_i to ATP) was reduced by a factor greater than 6 over control cultures. Impairment of both of these processes by Pb could ultimately influence bone growth and development.

Lead has also been show to perturb Epidermal Growth Factor's (EGF) control of intracellular calcium metabolism and collagen production in ROS cells (Long and Rosen, 1992). EGF is known to activate protein kinase C (PKC), resulting in increased calcium influx and through this mechanism, decreased collagen synthesis. Incubation of ROS cells with 5 micromolar Pb and 50 ng/mL EGF for 20 hours resulted in a 50% increase in total cell calcium versus the calcium increase seen in cells treated with EGF alone, suggesting more than one site of action is involved in calcium messenger perturbation. A similar finding was reported by Long and coworkers (1992) who found that treatment of Pb (25 micromolar) intoxicated osteosarcoma cells with parathyroid hormone (PTH, 400 mg/mL) resulted in a greater increase in cell calcium than with either treatment alone. Supplementary inhibition of collagen synthesis has also been reported with the addition of 25 micromolar Pb plus 50 ng/mL EGF, suggesting more than one site of action for the effect of Pb on collagen synthesis (Long and Rosen, 1992). Additional study has since suggested that Pb activates PKC in ROS cells and that PKC mediates the rise in intracellular calcium (Schanne et al., 1997). The observation that calphostin C, an inhibitor of PKC, prevented the Pb-induced elevation of intracellular calcium supported this hypothesis, as did the fact that free Pb at concentrations of 10^{-11} to 10^{-7} M directly activated PKC in the absence of activating concentrations of calcium. This would suggest Pb is capable of activating PKC at concentrations approximately 3,000 times lower than calcium.

Finally, Pb has been shown to be capable of inhibiting secretion of osteonectin, a bone related protein found in areas of active morphogenesis (Sauk et al., 1992). Treatment of ROS 17/2.8 cells with lead (4.5×10^{-6} M to $4.5\ 10^{-7}$ M) demonstrated that intracellular osteonectin levels were actually enhanced; however, the secretion of osteonectin into the media was delayed or inhibited. Protein production of collagen and the endoplasmic reticulum protein, Asp47, were relatively unaffected by Pb at these concentrations. The intracellular retention of osteonectin coincided with a decrease in levels of osteonectin mRNA, suggesting the processes associated with translation and secretion of osteonectin are sensitive to Pb.

5.8.5.4 Human Osteosarcoma Cells (HOS TE 85)

Evidence exists that Pb is directly osteotoxic to bone cells in culture. Studies examining the sensitivity of human osteosarcoma cells (HOS TE 85) to Pb found proliferation of the cells was inhibited at Pb concentrations of 4 μmol/l, while cytotoxicity occurred at the 20 μmol/l Pb concentration (Angle et al., 1993). In parallel experiments, rat osteosarcoma cells (ROS 17/2.8) were found to be somewhat less sensitive to the effects of Pb with inhibition of proliferation occurring at 6 μmol/l Pb concentration and cytotoxicity at Pb concentrations over 20 μmol/l.

5.8.5.5 Chick Chondrocytes

The effects of Pb on cartilage biology have been examined in isolated avian chondrocytes obtained from 3 to 5 week old chicks (Hicks et al., 1996). Exposure to media containing 0.1 to 200 μM Pb acetate or chloride were found to decrease thymidine incorporation, suppress alkaline phosphatase, and suppress both type II and type X collagen expression at the mRNA and protein levels. Cytotoxicity of the cultures from Pb exposure was dismissed as proteoglycan synthesis was found to be augmented, suggested Pb selectively inhibits specific aspects of the chondrocyte growth plate. Using the avian chondrocyte model, Zuscik et al. (2002) similarly reported Pb exposure (1 to 30 μM) causing a dose-dependent inhibition of thymidine incorporation into the growth plate, with a 60% reduction in proliferation at the highest concentration. Addition of TGF-β1 and PTHrP, regulators of growth plate, both separately stimulated thymidine incorporation, an effect that was dose-dependently blunted in the presence of Pb. At the highest Pb concentration (30 μM), inhibition was significantly less in the chondrocytes treated with Pb + TGF-β1 (24%) and Pb + PTHrP (19%) than for Pb alone (60%), suggesting the interaction of Pb with these growth factors may be independent of its primary action on the chondrocyte cells. Support for a direct action of Pb on these growth regulators is supported by the finding that normal TGF-β1 and PTHrP suppression of type X collagen expression is significantly reversed in a dose-dependent fashion in the presence of Pb. This effect evidently was not mediated by BMP-6 (Bone Morphogenic Protein), an inducer of terminal differentiation known to partially reverse the inhibitory effect of PTHrP, because in the presence of Pb, PTHrP significantly suppressed BMP expression, while combined exposure to Pb and TGF-β1 increased BMP expression approximately 3-fold. Further experiments performed on chick sternal chondrocyte cultures, utilized PTHrP responsive (AP-1) and non-responsive (NF-κB) reporter constructs to

examine potential effects of Pb on signaling. While having no effect on the basal activity of the AP-1 reporter, Pb dose-dependently enhanced PTHrP induction of the responsive AP-1 reporter. Lead dose-dependently inhibited the basal activity of the non-PTHrP responsive, NF-κB reporter. Taken together, these studies demonstrate that Pb has an inhibitory effect on the process of endochondral bone formation and that the effects of Pb are likely from its modulation of growth factors and second messengers involved in cell signaling responses.

5.8.6 Bone Lead as a Potential Source of Toxicity in Altered Metabolic Conditions

Lead is avidly taken up by bone and incorporated into bone matrix, where a substantial amount can remain over the lifetime of an organism. The uptake and incorporation of Pb into bone during acute exogenous exposures may be of short term benefit by limiting the exposure of other, more sensitive tissues; however, this does not eliminate Pb from the system. Subsequent release of Pb from this endogenous storage can produce a lifetime of steady, low level Pb exposure during periods of normal bone remodeling, while elevated Pb release during times of increased bone metabolism and turnover (i.e., pregnancy, lactation, menopause, and osteoporosis) can elevate blood levels of Pb significantly, potentially to toxic concentrations. This is especially relevant when there is concurrent exogenous exposure to Pb, as current blood Pb levels are a composite of current and past Pb exposure. Of greater concern is the mobilization of Pb during pregnancy and subsequent transfer to the developing brain of the fetus across the poorly developed blood:brain barrier. Maternal Pb also appears in breast milk, providing further exposure of the infant to Pb during lactation. Currently, the majority of animal studies examining mobilization of Pb from bone stores have focused principally on elevation of Pb levels or transfer of Pb, rather than reporting toxic effects associated with these exposures. Note that in most instances the mobilization and elimination of Pb is much faster in laboratory animals than in humans. For example, as discussed in Section 5.8.3, Hac and Kruchniak (1996) reported approximately 64% of Pb given over a 12 week period remained in the bones of rats 64 days post exposure. Therefore, the caveats of experiments performed in small animals, especially when examining mobilization of Pb stores, must be taken into consideration.

5.8.6.1 Pregnancy and Lactation

Pregnancy, and to a much greater extent, lactation, place significant calcium demands on the mother as she provides all the necessary calcium requirements of the developing fetus/infant. During these times of metabolic stress, increased demineralization of maternal bone occurs to supplement demand, unfortunately accompanied by the concurrent mobilization and release of Pb stored in the maternal skeleton from past exposure. Studies in several animal models have shown that maternal bone Pb can be mobilized during pregnancy and lactation, ultimately being transferred to the fetus during gestation and breast feeding. Keller and Doherty (1980) administered radiolabeled Pb drinking water (200μg/mL) to female mice for 105 days prior to mating or 105 days prior to mating and during periods of gestation and lactation (total 160 days of exposure). The results suggested very little Pb was transferred from mother to fetus during gestation, however, Pb transferred in milk and retained by the pups accounted for 3% of the maternal body burden of those mice exposed to Pb prior to mating only. The amount of Pb retained in these pups exceeded that retained in the mothers, suggesting lactation effectively transfers Pb burden from mother to suckling offspring. Transfer of Pb from mothers was significantly higher when Pb was supplied continuously in drinking water, rather than terminated prior to mating. Considerably higher lactational transfer of Pb from rat dams compared to placental transfer has also been reported (Palminger Hallén et al., 1996). Continuous exposure of rat dams to Pb until day 15 of lactation resulted in milk Pb levels 2.5 times higher than in whole blood, while termination of maternal Pb exposure at parturition yielded equivalent blood and milk levels of Pb, principally from Pb mobilized from maternal bone.

Using rats chronically exposed to Pb in drinking water, Maldonado-Vega et al. (1996) studied intestinal absorption of Pb, its mobilization, and redistribution during lactation. In rats exposed to Pb 144 days prior to lactation, the process of lactation itself elevated blood Pb and decreased bone Pb, indicating mobilization of Pb from bone as there was no external source of Pb during the lactation process. Rats exposed to Pb for 158 days (144 days prior to lactation and 14 days during lactation) also experienced elevated BLLs and loss of Pb from bone. Lead exposure only during the 14 days of lactation was found to significantly increase intestinal absorption and deposition (17 fold increase) of Pb into bone compared to non-pregnant rats, suggesting enhanced absorption of Pb takes place during lactation. As in other previous studies, the highest concentration of Pb in bone was found in non-pregnant non-lactating control animals,

with significantly decreased bone Pb in lactating rats secondary to bone mobilization and transfer via milk to suckling offspring. Follow-up studies examining the influence of dietary calcium found when calcium was altered from the normal 1% to 0.05%, bone calcium concentration decreased by 15% and bone Pb concentration decreased by 30% during the first 14 days of lactation (Maldonado-Vega et al., 2002). In non-lactating rats on the 0.05% calcium diet, there were also decreases in bone calcium, but neither incremental bone resorption nor Pb efflux from bone, suggesting the efflux from bone during lactation was related to bone resorption. Of interest, enhancement of calcium (2.5%) in the diet of lactating rats increased calcium concentration in bone by 21%, but did not decrease bone resorption, resulting in a 28% decrease in bone Pb concentration and concomitant rise in systemic toxicity. In both studies, the authors concluded that Pb stored in bone should be considered a major source of self-intoxication and of exposure to suckling offspring.

In one of few studies showing a toxic effect, Han et al. (2000) demonstrated adverse effects in rat offspring born to females whose exposure to Pb ended well before pregnancy. Five week-old-female rats had been given Pb-acetate in drinking water (250 mg/mL) for five weeks, followed by a one month period without Pb exposure before mating. To test the influence of dietary calcium on Pb absorption and accumulation, some pregnant rats were fed diets deficient in calcium (0.1%) while others were maintained on a normal calcium (0.5%) diet. As expected, all Pb-exposed dams and pups had elevated blood Pb levels; however, pups born to dams fed the diet deficient in calcium during pregnancy had higher blood and organ Pb concentrations compared to pups from dams fed the normal diet. Significantly, pups born to Pb-exposed dams had lower mean birth weights and birth lengths than pups born to non-Pb-exposed control dams ($p < 0.0001$), even after confounders such as litter size, pup sex, and dam weight gain were taken into account. The authors concluded that while increases in dietary calcium during pregnancy are capable of reducing Pb accumulation in the fetus, they cannot prevent the decreases in birth weight and length associated with pre-maternal Pb exposure and subsequent mobilization. This has relevance in human pregnancy, as many women experience exposure to Pb during their lifetimes (especially during childhood) and mobilization of the Pb from bone stores during pregnancy could present toxic complications.

Within the last decade, an invaluable method to explore the kinetics of Pb transfer from bone to blood has been developed and evaluated (Inskip et al., 1996; O'Flaherty et al., 1998).

The method utilizes recent administration of sequential doses of Pb mixes enriched in stable isotopes (^{204}Pb, ^{206}Pb, and ^{207}Pb) to female cynomolgus monkeys (*Macaca fascicularis*) that have been chronically (1,300 to 1,500 µg Pb/kg body weight per day for ten years or greater) administered a common Pb isotope mix. The stable isotope mixes serve as a marker of recent, exogenous Pb exposure, while the chronically administered common Pb serves as a marker of endogenous (principally bone) Pb. From thermal ionization mass spectrometry analysis of the Pb isotopic ratios of blood and bone biopsies collected at each isotope change, and using end-member unmixing equations, it was determined that administration of the first isotope label allows measurement of the contribution of historic bone stores to blood Pb. Exposure to subsequent isotopic labels allowed measurements of the contribution from historic bone Pb stores and the recently administered enriched isotopes that incorporated into bone (Inskip et al., 1996). In general the contribution from the historic bone Pb (common Pb) to blood lead level was constant (~20%), accentuated with spikes in total blood Pb due to the current administration of the stable isotopes. After cessation of each sequential administration, the concentration of the signature dose rapidly decreased. Initial attempts to apply a single-bone physiologically based model of Pb kinetics were unsuccessful until adequate explanation of these rapid drops in stable isotopes in the blood were incorporated (O'Flaherty et al., 1998). Once revisions were added to account for rapid turnover of the trabecular bone compartment and slower turnover rates of cortical bone compartment, an acceptable model evolved. From this model it was reported that historic bone Pb from 11 years of continuous exposure contributes approximately 17% of the blood Pb concentration at Pb concentration over 50 µg/dL, reinforcing the concept that the length of Pb exposure and the rates of past and current Pb exposures help determine the fractional contribution of bone Pb to total blood Pb levels (O'Flaherty et al., 1998). The turnover rate for cortical (~88% of total bone by volume) bone in the adult cynomolgus monkey was estimated by the model to be ~4.5% per year, while the turnover rate for trabecular bone was estimated to be 33% per year.

Using the method of sequential stable isotope administration, Franklin et al. (1997) examined flux of Pb from maternal bone during pregnancy of 5 female cynomolgus monkeys who had been previously exposed to common Pb (approximately 1,100 to1,300 µg Pb/kg body weight) for about 14 years. In general, lead levels in maternal blood attributable to Pb from mobilized bone were reported to drop 29 to 56% below prepregnancy baseline levels during the

first trimester of pregnancy. This was ascribed to the known increase in maternal fluid volume, specific organ enlargement (e.g., mammary glands, uterus, placenta), and increased metabolic activity that occurs during pregnancy. During the second and third trimesters, when there is a rapid growth in the fetal skeleton and compensatory demand for calcium from the maternal blood, the Pb levels increased up to 44% over pre-pregnancy levels. With the exception of one monkey, blood Pb concentrations in the fetus corresponded to those found in the mothers, both in total Pb concentration and proportion of Pb attributable to each isotopic signature dose (common = 22.1% vs. 23.7%, ^{204}Pb = 6.9% vs. 7.4%, and ^{206}Pb = 71.0% vs. 68.9%, respectively). From 7 to 25% of the Pb found in fetal bone originated from maternal bone, with the balance derived from oral dosing of the mothers with isotope during pregnancy. Of interest, in offspring from a low Pb exposure control monkey (blood Pb <5 μg/100 g) ~39% of Pb found in fetal bone was of maternal origin, suggesting enhanced transfer and retention of Pb under low Pb conditions.

Clearly, the results of these studies show that Pb stored in bone is mobilized during pregnancy and lactation, exposing both mother and fetus/nursing infant to blood/milk Pb levels of potential toxicity. Of equal concern, a significant proportion of Pb transferred from the mother is incorporated into the developing skeletal system of the offspring, where it can serve as a continuing source of toxic exposure. The above study by Franklin et al. (1997) illustrates the utility of sequentially administered stable isotopes in pregnancy; however, its use may also be applicable in studies of lactation, menopause, osteoporosis, and other disease states where mobilization of bone and release of Pb stores occurs. Furthermore, given that isotopic ratios of common Pbs vary by location and source of exposure, when humans migrate from one area and source of exposure to another, it is possible to document changes in mobilized Pb, especially during times of metabolic stress.

5.8.6.2 Age/Osteoporosis

The age of an animal at the time of exposure to Pb has been shown to influence the uptake and retention of Pb by bone. In experiments to determine the influence of age on this process, Han et al. (1997) exposed rats for five weeks to 250 mg/L Pb-acetate in drinking water beginning at 5 weeks of age (young child), 10 weeks of age (mid-adolescence), or 15 weeks of age (young adult), followed by a 4 week period of without Pb exposure. An additional group of rats were exposed to Pb beginning at 5 weeks, but examined following an 8 or 20 week period after

cessation of Pb. Significantly lower blood and bone Pb concentrations were associated with greater age at the start of Pb exposure and increased interval since the end of exposure. However, young rats beginning exposure to Pb at 5 weeks and examined 20 weeks after cessation of exposure, still had bone Pb concentrations higher than those found in older rats only 4 weeks after cessation of exposure. This demonstrated that exposure to Pb at a young age leads to significant skeletal Pb accumulation and retention, despite the high rate of bone remodeling that occurs during growth and development at that time.

At the opposite end of the spectrum, Cory-Slechta et al. (1989) studied differences in tissue distribution of Pb in adult and old rats. Adult (8 months old) and old (16 months old) rats were exposed to 50 ppm Pb-acetate in drinking water for 11 months, at which time the experiment was completed. Bone (femur) Pb levels in older rats were found to be less than those in younger rats; however, blood lead levels were higher in the older rats. Of interest, brain Pb concentrations in the older rats exposed to Pb were significantly higher, and brain weights were significantly less than the brain Pb concentration and weights of unexposed older control rats or adult rats exposed to Pb, suggesting a potential detrimental effect. The authors suggested that a possibility for the observed differences in tissue concentrations of Pb was due to changes in the capacity of bone to store Pb with advanced age. In a subsequent study, Cory-Slechta (1990b) examined kinetic and biochemical responses of young (21 day old), adult (8 months old), and old (16 months old) rats exposed to Pb at 0, 2, or 10 mg Pb acetate/kg/day over a 9.5 month experimental period. Results suggested that older rats may have increased vulnerability to Pb due to increased exposure of tissues to Pb and greater sensitivity of the tissues to the effects of Pb. As in the previous study (Cory-Slechta et al., 1989), lower bone levels of Pb were present in older rats with concomitant elevated levels of Pb in brain and other tissues, supporting the hypothesis that exposure to Pb over a lifetime may contribute to deterioration of health in old age, potentially during times of heightened bone remodeling such as occurs during osteoporosis. In studies of bone Pb metabolism in a geriatric, female nonhuman primates exposed to Pb approximately 10 years previously, McNeill et al. (1997) reported no significant changes in bone Pb level over a 10 month observation period as measured by ^{109}Cd K X-ray fluorescence. The mean half-life of Pb in bone of these animals was found to be 3.0 ± 1.0 years, consistent with data found in humans, while the endogenous exposure level due to mobilized Pb was 0.09 ± 0.02 μg/dL blood. Results examining Pb accumulation in the bones of aging male mice suggest

low levels of bone Pb contributing to the osteopenia observed normally in C57BL/6J mice (Massie and Aiello, 1992). The mice were maintained on regular diet (0.258 ppm Pb) and water (5.45 ppb Pb) from 76 to 958 days of age. While the Pb content of femurs increased by 83%, no significant relationship was found between Pb and bone density, bone collagen, or loss of calcium from bone.

5.8.6.3 Weight Loss

The relationship between body mass and bone mass is highly correlated and during times of loss of body weight, such as dietary restriction, a concomitant loss of bone mass also occurs. It is therefore possible that Pb stored in bone from prior exposures could be released into the system as skeletal bone is mobilized and result in Pb toxicity. To examine the influence of weight loss on release of stored Pb, Han et al. (1996) first exposed rats to Pb in drinking water (250 mg/l of Pb as acetate) for 5 weeks, followed by a 4 week washout period without Pb to allow primarily accumulation in the skeleton. Rats were then randomly assigned to a weight maintenance group, a moderate weight loss group (70% of maintenance diet), or a substantial weight loss group (40% of maintenance diet) for a four week period. At the end of this experimental period the blood and bone levels of Pb did not differ between groups, however, the amount and concentration of Pb in the liver increased significantly. A follow up study in rats previous exposed to Pb for two weeks was undertaken to determine the effect of weight loss and exercise on the distribution of Pb (Han et al., 1999). They found weight loss secondary to dietary restriction to be the critical factor elevating organ Pb levels and, contrary to their first study, elevated blood levels of Pb. No significant difference in organ or blood Pb concentrations were reported between the exercise vs. no exercise groups. These studies suggest Pb toxicity could occur in those previously exposed to Pb during times of dietary restriction.

5.8.7 Bone and Lead Summary

Lead substitutes for calcium and is readily taken up and stored in the bone of experimental animals, potentially allowing bone cell function to be compromised both directly and indirectly by exposure. In general, relatively short term exposure of mature animals to Pb does not result in significant growth suppression, however, chronic Pb exposure during times of inadequate nutrition have been shown to adversely influence bone growth, including decreased bone density,

decreased trabecular bone, and growth plates. Exposure of developing animals to Pb during gestation and the immediate postnatal period has clearly been shown to significantly depress early bone growth in a dose-dependent fashion, though this effect is not manifest below a certain threshold. Numerous mechanisms for the toxic effect of Pb on bone have been explored using various animal models. Systemically, Pb has been shown to disrupt mineralization of bone during growth, to alter calcium binding proteins, and to increase calcium and phosphorus concentration in the blood stream, in addition to potentially altering bone cell differentiation and function by altering plasma levels of growth hormone and calciotropic hormones such as vitamin D_3 [1,25-$(OH_2)D_3$].

Bone cell cultures of both animal and human derivation have substantially contributed to the general understanding of the adverse effects of Pb on bone cell metabolism directly and its indirect effect on bone and bone cells by perturbation of numerous local and systemic factors. These in vitro studies have indicated that Pb is primarily taken up by osteoclasts and likely perturbs intracellular calcium homeostasis secondary to osteoclastic bone resorption. Bone cell proliferation is also inhibited. Exposure of bone cell cultures to Pb has been shown to impair vitamin D-stimulated production of osteocalcin, inhibit secretion of bone-related proteins such as osteonectin and collagen, and suppress bone cell proliferation, potentially by interference with such factors as GH, EGF, TGF-β1, and PTHrP.

Finally, several animal studies have suggested Pb stored in bone can serve as a continuing, endogenous source of exposure for an individual or can be transferred from mother to offspring during pregnancy and/or lactation, with potentially toxic consequences. Periods of extensive bone remodeling, (i.e., during weight loss, advanced age, altered metabolic state, and pregnancy and lactation) are all associated with mobilization of Pb stores from bone of animals. During pregnancy, transfer of Pb from mother to offspring has been documented, however, available evidence suggests a more significant transfer from mother to offspring occurs during lactation when the concentration of Pb in mother's milk can be several times higher than corresponding blood levels. Despite the extensive remodeling of bone that occurs during growth and development of young animals, a significant amount of Pb can be accumulated and retained during times of exposure.

5.8.8 Teeth – Introduction

There was little information in the prior 1986 AQCD relating lead exposure to adverse outcomes in the teeth of animals. At that time, the incorporation of Pb into teeth was recognized as was the fact that tooth Pb increased with age, proportional to the rate of exposure and roughly proportional to the blood Pb concentration.

Teeth consist of a hard outer layer of enamel, supported by an underlying layer of dentin, which itself is supported by a connective tissue known as the dental pulp. Enamel is the hardest substance in the body and the most highly mineralized, consisting of ~96% mineral (calcium hydroxyapatite substituted with carbonate ions) and 4% other organic materials, while dentin is only ~70% mineral. The formation of enamel (amelogenesis) occurs as a two stage process of organic matrix production with ~30% mineralization, followed by removal of water and proteins from the matrix with concurrent further mineralization. As in bone, Pb ions are apparently capable of substituting for calcium ions in the mineralizing tooth, becoming essentially trapped. However, unlike bone, the tooth, with subtle exceptions, does not undergo a remodeling process. Dentin formation (dentinogenesis) can be likened to endochondral bone formation, in that an unmineralized matrix (predentin, rather than cartilage) is laid down first, followed by mineralization to mature dentin. The cells responsible for amelogenesis and dentinogenesis, called ameloblasts and odontoblasts respectively, are similar to osteoblasts in that they respond to various signaling factors, secrete matrix proteins, and create an environment favorable to deposition of minerals. After enamel formation on a specific tooth is completed, ameloblasts are lost and no additional enamel is laid down with the exception of certain teeth in rodents. These teeth, typically incisors on rats, mice, and most other rodents, continuously erupt to offset the attrition that occurs with daily use. Therefore, the process of amelogenesis is ongoing, albeit confined to a localized area, throughout the life of the animals. For this reason rodents have been utilized extensively to examine the processes of amelogenesis and the influence of various toxic agents, such as Pb, on tooth development. Ameloblasts are especially sensitive to toxins and altered metabolic conditions and respond to such insults with disruption of enamel formation. When disruption occurs, defects in the enamel can occur, typically as a band of malformed or altered enamel. As described below, exposure of animals to various concentrations of Pb during tooth development is not only capable of creating distinctive marking of enamel ("Pb lines"), but may influence the resistance of the enamel to dental decay.

Within the dental pulp, a layer of odontoblasts continue to reside against the inner layer of the primary dentin for the life of the tooth. During this time the odontoblasts are systematically slowly putting down thin layers of secondary dentin, slowly decreasing the size of the pulp chamber with age. Lead present during this process has been shown to be readily taken up by this dentin layer, providing a potential marker of historic Pb exposure. Though the enamel is a non-living substance, it is not entirely inert. The external surface of enamel is more or less in a continuous state of flux or turnover as it chemically demineralizes from acids consumed or produced in the mouth by bacteria, followed by remineralization of demineralized enamel when contact with saliva supersaturated with calcium and phosphate ions occurs. Lead present during this process can easily be released from enamel and/or incorporated initially or back into it depending on the circumstances.

In summary, Pb has the potential to disrupt the various processes associated with formation of teeth, plus incorporate itself into all mineralized tooth tissues during formation. Posteruptively, Pb can become incorporated into the secondary dentin, and can be taken up or released from the outer surface layer of enamel during times of remineralization/ demineralization. As described below, exposure of animals to Pb has been associated with adverse dental outcomes.

5.8.9 Uptake of Lead by Teeth

As seen with bone, uptake of Pb into the teeth of animals has been demonstrated in a number of studies and by multiple routes of administration. Twenty four hours after a single intraperitoneal injection of radioactive Pb-203 (^{203}Pb, 1 μg/kg) to young (15 day suckling rats) and old (120 day) female rats, 0.7% of the injected dose was present in the four incisor teeth of the younger animals and 0.6% was present in the same teeth of the older animals (Momcilovic and Kostial, 1974). These percentages jumped to 1.43% and 0.88%, respectively, 192 hours after the injection, suggesting incorporation and retention of Pb by teeth is greater in younger animals than in adults, as found in bone. Lead has also been shown to be incorporated into incisors of rats exposed to airborne Pb. Grobler and coworkers (1991) exposed 6 week old rats to either "Clean Air" (0.05 μg Pb/m^3) or air containing 77 μg Pb/m^3 and found significant differences in the amount of Pb incorporated into the incisors of the animals. After 70 days, a mean of only 0.8 μg Pb/g of incisor dry mass was found in incisors from control animals, while

11.0 µg Pb/g was present in incisors from the 77 µg Pb/m^3 group. Exposure to air containing 249 µg Pb/m^3 for 28 days or to 1,546 µg Pb/m^3 for 50 days resulted in mean values of 13.8 and 153 µg Pb/g incisor dry weight of Pb incorporation, respectively, highlighting the fact that dose and length of exposure are determinates of amount of Pb contained in the teeth of these animals. Lead has also been shown to be taken up into the teeth of weanling rats whose mothers were exposed to Pb in drinking water. The offspring of pregnant rats exposed during gestation and lactation until 21 days post partum to water containing 0, 3, or 10 ppm Pb showed dose-dependent, significant increases in the Pb content of incisors, first molars, and second molars (Grobler et al., 1985). Taken together, these studies confirm the uptake of Pb into teeth as delivered by various means and suggest that maternal exposure can result in uptake in offspring, during gestation and/or lactation.

5.8.10 Effects of Lead on Enamel and Dentine Formation

Early microscopic studies by Eisenmann and Yaeger (1969) confirmed alterations in rat incisor enamel formation 7 days after a single SC dose of Pb (0.15 or 1.5 mM/100g animal weight); however, no effect was seen at the 0.075 mM/100g dose. Lead was found to have inhibited mineralization of both enamel and dentin, but only to a "mild to moderate" extent with the mineralization of dentin more affected. It was speculated at the time that Pb could affect the production of normal, mineralizable organic matrix; affect enzymes specific to enamel or dentin formation; affect crystal structure and/or growth; or affect a combination of these factors. In studies of dentinogenesis, incubation of fixed rat molar germs with Pb-pyrophosphate has shown localization of Pb to the mineralization front of dentin (i.e., the area of recently formed dentin), to the stratum intermedium, and to subodontoblastic cells, suggesting Pb may react with mineral components located in the mineralization zone or have a high affinity for these incompletely mineralized areas (Larsson and Helander, 1974). Localization of Pb was also seen at the area of the dentino-enamel junction. Similar examination of first molar germs from 3-day-old rats showed that Pb also localized to the periphery of dentinal globules (Larsson, 1974). A single injection of Pb-acetate (30 mg/kg body weight) produces an immediate (within 6 h) response in the growing dentin of the rat incisor, leading to the formation of a so-called "Pb line" (Appleton, 1991). A transient rise in serum calcium and phosphorus accompanied the injection, leading to speculation that lead may have been replacing these minerals in the apatite structure. However,

backscattered electron imaging of the Pb line showed it to be composed of continuous hypomineralized interglobular dentin with some incomplete fusion of calcospherites resulting in uneven mineralization, but no localized concentration of Pb was detectable. This is consistent with Featherstone and co-workers (1981) who reported that Pb incorporation during apatite synthesis was widely dispersed, rather than concentrated in areas of calcium deficiency. Once synthesis is complete, however, Pb is capable of entering calcium deficient areas in enamel, substituting for calcium (Featherstone et al., 1979). This is essentially the process that occurs during demineralization/remineralization of enamel. Appleton (1991, 1992) suggested that Pb has a direct effect on odontoblasts, creating a local disturbance of calcium metabolism, a process similar to that described in bone (Pounds et al., 1991). Interestingly, no ultrastructural changes in ameloblasts from rat pups whose mothers had been drinking water containing Pb was observed.

During the normal process of amelogenesis, water and proteins contained within the organic matrix are lost, leaving densely mineralized enamel. The removal of enamel proteins during this phase is facilitated by enamel proteinases, which are believed to degrade the proteins into smaller units capable of diffusing from the matrix. Using crude extracts from scrapings of rat incisor teeth, Gerlach and co-workers (2000a) demonstrated that Pb inhibited these proteinases in vitro at micromolar concentrations. In rats given drinking water containing Pb at either 0, 34, or 170 mg/L as Pb-acetate for 70 days, increased amounts of proteins were found in enamel matrix from animals exposed to Pb (Gerlach et al., 2002). Moreover, enamel microhardness analysis of upper incisors revealed a significant decrease in microhardness in regions of enamel maturation, but not in areas of fully mature enamel, suggesting Pb exposure mediates a delay in enamel mineralization. In adult rats with incisors trimmed to remove occlusal (biting) contact, a single IP dose of Pb-acetate (40 mg/kg) significantly delayed the continuous eruption of the incisor at all time points between 8 and 28 days after dosing, compared with controls (Gerlach et al., 2000b). It is of interest that delayed eruption of teeth in children living in areas of heavy metal contamination (Pb and zinc) has been reported previously (Curzon and Bibby, 1970).

5.8.11 Effects of Lead on Dental Pulp Cells

Hampered by a general lack of cell cultures specifically for teeth, there remains a paucity of information regarding both the cultures themselves and the effect of Pb upon such cultures. In a single in vitro study using a human dental pulp cell culture obtained from teeth extracted for orthodontic purposes, Thaweboon and co-workers (2002) examined the effects of three concentrations (4.5×10^{-5} M, 4.5×10^{-6} M, 4.5×10^{-7} M) of Pb-glutamate on cell proliferation, protein production, and osteocalcin secretion. Under serum free conditions (DMEM only) all concentrations of Pb significantly increased cell proliferation on day 1, day 3 and day 5 of exposure, as measured indirectly by mitochondrial dehydrogenase enzyme assay. In the presence of 2% fetal bovine serum only, the higher concentration of Pb significantly increased protein production, suggesting an influence of serum constituents on cell growth or binding of free Pb in the medium. Similar results were reported when rat osteosarcoma cells (ROS 17/2.8) were exposed to identical concentrations of Pb over 2-, 4-, and 6-day time points (Sauk et al., 1992). Concentrations of Pb less than 4.5×10^{-5} M concentration did not affect osteosarcoma cell proliferation in the presence of serum, but in the absence of serum 4.5×10^{-7} M Pb increased cell proliferation at day 4, while at day 6, 4.5×10^{-6} M Pb inhibited proliferation. Further testing of human dental pulp cells in serum-free conditions showed that Pb exposure caused dose-dependent decreases in intracellular protein and procollagen type I production over the 5-day period experimental period (Thaweboon et al., 2002). Short-term exposure of the cells to Pb significantly decreased osteocalcin production in a dose-dependent manner at 8- and 12-h exposure time points. These results suggest that Pb is capable of exerting multiple toxic effects on cells derived from human dental pulp.

5.8.12 Adverse Effects of Lead on Teeth—Dental Caries

In a recent review, Bowen (2001) highlighted 12 epidemiological studies that examined the association between Pb exposure and dental caries (decay), reporting that 8 studies supported the concept that Pb is a caries-promoting element. Unfortunately, the source and actual exposure to Pb and measurement of prevalence of caries varied greatly, providing less than completely satisfactory evidence in the opinion of the author. There is also a paucity of well-controlled animal studies examining this issue.

In an early study examining the effect of drinking solutions containing various metallic ions on dental caries in hamsters, Wisotzky and Hein (1958) reported post-eruptive ingestion of drinking water containing 0.5 mEq of Pb significantly increased caries scores in molar teeth of males after 84 days, but, perplexingly, not in females after 98 days of exposure. It should be noted that in animal studies such as these it is routine to maintain the animals on cariogenic or caries-promoting diets high in fermentable sugars. Clear evidence supporting Pb's role in enhancing susceptibility to dental caries was reported by Watson and co-workers in 1997. In their study, female rats were exposed to Pb in drinking water (34 ppm as Pb-acetate) as young adults, during pregnancy, and during lactation. Lead exposure of the subsequent offspring from the dams was, therefore, from transfer of endogenous Pb from dam to pup during gestation and lactation, with no further exposure after weaning. This pre- and perinatal exposure to Pb resulted in a significant, almost 40%, increase in the prevalence of dental caries over control animals. The study was significant for other reasons, as it mimicked the conditions found in many inner cities where young females are exposed to Pb in their environment and later transfer this Pb to their own fetuses during the extensive bone remodeling that occurs during pregnancy and lactation. The mean blood Pb level in the dams upon weaning was 48 μg/dL, which is not unlike upper levels reported in humans.

The mechanisms by which Pb enhances susceptibility to caries remain uncertain, though clearly altered mineralization and/or incorporation of Pb into enamel as described above could enhance its solubility in acid. Lead also appears in the saliva of rats at about 5% of the whole blood level and at about 61% of the plasma filtrate Pb level (Mobarak and P'an, 1984), providing an avenue for post-eruptive interaction with the exposed enamel in the oral cavity. Notably, decreased salivary flow has been reported in rats exposed to Pb, and decreased salivary function is known to increase caries risk. Stimulated parotid function was decreased by nearly 30% in the Pb-exposed offspring in the study by Watson and co-workers (1997), an effect that could have been mediated by the salivary gland requirement of intact parasympathetic and sympathetic nervous systems for normal development (Schneyer and Hall, 1970) and Pb's known adverse effect on neurotransmitters (Bressler and Goldstein, 1991). Acute infusion of 4 μg of Pb per min has been reported to significantly reduce pilocarpine-stimulated salivary secretion in rats over a 50-min period (Craan et al., 1984), while 24-day administration of 0.05% Pb-acetate significantly reduced the concentration of protein and calcium in pilocarpine-stimulated rat

submandibular saliva (Abdollahi et al., 1997). Of potential interest, postnatal exposure of rats to Pb (10 or 25 ppm in drinking water) and a caries-enhancing diet containing fluoride (sucrose containing 15 ppm fluoride) was not associated with an increased risk of dental caries, suggesting that Pb does not interfere with the protective effect of fluoride (Tabchoury et al., 1999). Clearly though, the effect of Pb exposure on salivary gland function and the mechanism by which Pb exposure enhances caries risk needs to be further explored.

5.8.13 Lead from Teeth as a Potential Source of Toxicity

Although no studies currently document the contribution of Pb incorporated into teeth as a source of endogenous Pb exposure, the potential exists during the process of exfoliation of the primary dentition. As described above (Section 5.8.9) Pb is avidly incorporated into the developing dentin and enamel components of teeth. Like bone, the uptake and incorporation of Pb into teeth during acute exogenous exposures may be of short-term benefit by limiting the exposure of other, more sensitive tissues, but, unlike bone, teeth do not undergo a gross remodeling process (the continuous, superficial demineralization/remineralization of the exposed tooth surfaces, principally enamel, are assumed here to be insignificant). However, during the exfoliative process, the erupting secondary tooth erodes away the root (composed of cementum and dentin) of the overlying primary tooth along with some surrounding alveolar bone. Any Pb incorporated into these portions of bone and primary tooth would be released by the erosive process, with the potential to produce highly elevated local concentrations of Pb in the proximity of remodeling alveolar bone and developing secondary teeth. A more modest contribution to circulating blood Pb would be predicted. Animal research in this area has been hampered, as most common rodents (i.e., rats, mice) are monophyodonts (have only one set of teeth). Although monkeys are an acceptable model, it is problematic how release of Pb stored in teeth could be differentiated from that of remodeling skeletal bones formed at a similar time point, plus the disproportionate size of the skeletal mass compared to the dentition may mask any contribution of Pb mobilized by exfoliation.

5.8.14 Teeth and Lead Summary

As found with bone, Pb substitutes for calcium and is readily taken up and incorporated into the developing teeth of experimental animals. Unlike bone, teeth do not undergo

remodeling per se and, with few exceptions, most Pb incorporated into tooth structure remains essentially in a state of permanent storage. Administration of high doses of Pb to animals has demonstrated the formation of a Pb line, visible in both the enamel and dentin and localized to areas of recently formed tooth structure. Within this Pb line, areas of inhibition of mineralization are evident in enamel and dentin. Lead has been shown to decrease cell proliferation and production of intracellular protein, procollagen type I, and osteocalcin in human dental pulp cells in culture. Studies of Pb exposure in adult rats have reported inhibition of post-eruptive enamel proteinases, delayed teeth eruption times, and decreased microhardness of surface enamel. During the process of enamel formation, Pb is apparently widely dispersed when first incorporated into the developing apatite crystal; however, post-formation, Pb is capable of entering and concentrating in calcium-deficient areas within the enamel. Whether Pb incorporation into the enamel surface compromises the integrity and resistance of the surface to dissolution, and ultimately increases risk of dental decay, is unclear. Numerous epidemiologic studies suggest Pb is a caries-promoting element. Animal studies (both post-eruptive Pb exposure and pre- and perinatal Pb exposure studies) support this concept, although the exact mechanism of action remains elusive. No animal studies have examined the role exfoliation of the primary dentition in release of Pb previously stored in tooth structure, though it is likely this process could serve as an additional source of Pb exposure in childhood.

5.9 EFFECTS OF LEAD ON THE IMMUNE SYSTEM

The immune system, along with the neurological system, has emerged as one of the more sensitive targets of Pb-induced toxicity. However, because Pb exposure at low to moderate levels does not produce overt cytotoxicity of immune cells, immune-associated health effects result from misregulation and shifts in functional capacity rather than profound lymphoid deficiencies. As a result, the most sensitive biomarkers of Pb-induced immunotoxicity are those associated with specific functional capacities as opposed to measures of cell enumeration and/or lymphoid organ pathology. This distinguishes Pb from some other types of immunotoxicants. The following sections provide a survey of the reported immune effects resulting from exposure to Pb in humans and animal models. In general, the focus is on those studies that have been reported since the 1986 AQCD (U.S. Environmental Protection Agency, 1986) was prepared and

have altered our understanding of lead-induced immunotoxicity and the associated immune-related health risks.

5.9.1 Introduction

The comparative development of the immune system in humans and animal models used for immunotoxicology was reviewed in recent years by Payne and Crooks (2002) and Holsapple et al. (2003). Pluripotent hematopoietic stem cells arise from uncommitted mesenchymal stem cells located in the spanochnopleure area near the heart and appear in the yolk sac (Holsapple et al., 2003). During human gestation, these stem cells first migrate at approximately 5 weeks and produce lymphoid and myeloid stem cells. Lymphoid stem cells can be found in the liver at approximately 7–8 weeks of gestation. In the mouse, the hematopoietic stem cells migrate to the liver on gestational day (GD) 10.

Migration of stem cells to the thymus occurs in humans about the 9th week of gestation and in mice on GD 11. The equivalent migration probably happens in the rat at GD 13 or later. Bone marrow lymphopoieis begins in humans about week 12 of gestation and in mice about GD 18. Immune development continues postnatally in humans as well as rodents. During embryonic development, immune maintenance of the pregnancy is important, and Th2 development is favored over Th1. Among other things, the capacity of dendritic cells to promote Th1 activity is dramatically suppressed in the newborn (Langrish et al., 2002). However, Th1 cytokines can be stimulated shortly after birth in humans (Malamitsi-Pichner et al., 2005). However, it is clear that, at birth, rodents lag behind in immune development compared with humans (Dietert et al., 2000; Holsapple et al., 2003).

Immune maturation continues in the thymus and bone marrow to give rise to the broad spectrum of myeloid and lymphoid cells that contribute to host defense and tissue homeostasis. The thymus-derived (T) lymphocytes provide regulatory cells facilitating a wide range of acquired immune responses and also produce cytotoxic T lymphocytes capable of attacking tumor and virally infected cells. Among regulatory T lymphocytes are at least two types of helper populations termed T helper 1 (Th1) and T helper 2 (Th2). The former regulatory cells promote immune responses helpful against intracellular pathogens, while the latter are important in defense against extracellular pathogens. However, skewing of the Th1/Th2 balance too far in

either direction is problematic in terms of health risk. Such skewing is a large factor in the consideration of lead-induced immunotoxicity (see Sections 5.9.2, 5.9.4, and 5.9.8).

B lymphocytes are named for the Bursa of Fabricius, an organ important in their development in avian species. They constitute the other major lymphoid cell type important in acquired immunity. B lymphocytes produce first membrane-bound and then secreted immunoglobulins (antibodies) that are a significant part of humoral immunity. Different classes of immunoglobulins produced during class switching, and promoted by different T lymphocyte cytokines, are tailored to be effective against different types of pathogens (e.g., viruses vs. extra-cellular parasites). The potential of Pb on B lymphocytes is discussed along with humoral immunity in Section 5.9.3.1.

Another lymphoid cell type is the natural killer (NK) cell. These cells function during innate immunity as a front line defense against tumor cells and virally infected cells. NK cells have the capacity to recognize a limited number of receptors on target cell surfaces, including the loss of Class I (major histocompatibility complex) proteins. Such self-protein identity loss is usually associated with viral infection of host cells. NK cells also produce cytokines capable of regulating macrophage and T cell activity and, in turn, NK cells can be activated by lymphoid-produced cytokines. Consideration of the effect of Pb on NK cells is presented in Section 5.9.8.

Myelomonocytic cells such as macrophages and polymorphonuclear leukocytes (neutrophils) are also important in innate immunity. Neutrophils are usually short-lived cells that are capable of leaving the circulation and migrating into tissues. From there they can phagocytize pathogenic targets, utilize phagolysosomes to destroy bacteria, and secrete significant quantities of reactive oxygen intermediates (ROIs) into the local environment. They are also capable of producing nitric oxide (NO) in most species. The impact of Pb on neutrophils is presented in Section 5.9.7.

Macrophages are much longer-lived and can perform many of the same functions as neutrophils. Unlike neutrophils, macrophages can inactivate much of the ROIs they produce internally. However, they can produce vast quantities of NO and have major functional roles in tissue homeostasis, lymphoid regulation, and antigen presentation. In fact, macrophages reside in virtually every tissue, although their morphology can vary widely and their function spectrum can be quite distinct among different specialized organs. Kupffer cells in the liver and alveolar

macrophages in the lungs are two examples of highly specialized forms of macrophages. Misregulated or misdirected macrophage activity is a major cause of immune-inflicted tissue damage. When the cells are activated within tissues and chronically overproduce proinflammatory cytokines such as interleukin-1 (IL-1), tumor necrosis factor-alpha (TNF-α), and interleukin-6 (IL-6) in addition to NO and ROIs, the result is usually tissue damage and loss of function, if not ultimately cancer. A classic example of the potentially destructive role of misregulated macrophages is found in the case of asbestos-induced pathology of the lung (Holian et al., 1997; Driscoll, 2000). Misregulation of macrophages is a major consideration in the case of Pb (discussed in Section 5.9.6).

Dendritic and Langerhans cells are related in lineage to macrophages. These cells are vitally important in antigen trafficking and presentation, particularly in lymph nodes and skin. However, far too little is known about their potential sensitivity to lead, largely because their isolation, complete phenotypic and functional characterizations, and inclusion in assessment methodologies are relatively recent developments within immunology.

The other major cell types important in a consideration of lead-induced immunotoxicity are basophils, eosinophils, and mast cells. Basophils are involved in various inflammatory reactions, but little is known about the direct effect of Pb on this cell population. Mast cells are fixed tissue cells surrounding the vasculature in many organs. These cells can secrete preformed highly vasoactive products with inflammatory potential (e.g., kinins and histamine) in response to cell-surface-initiated signals. Mast cells can be triggered by numerous signals including both substance P and immunoglobulin E (IgE). Because mast cell-induced inflammation is associated with IgE-mediated allergic reactions, these cells are important in the clinical ramification of lead-induced immunotoxicity. Eosinophils are a granulocytic cell type associated with Th2-driven inflammatory reactions. They frequently appear in association with allergic reactions and are regulated by numerous lymphoid cells as well as by mast cells. Remarkably little is known about the direct effects of Pb on eosinophil function despite the probable role of these cells in certain allergic manifestations following exposure to lead.

Life-stage related differences in immunotoxicological risk have been reviewed by several authors (Barnett 1996; Holladay and Smialowitz, 2000; Dietert et al., 2002; Holladay, 2005), and it seems clear that the vulnerability of the developing immune system to immunotoxic insult is significantly greater than that of the fully-matured and dispersed immune system for the vast

majority of immunotoxicants (Luebke et al., 2005). Dietert et al. (2000) and Holsapple et al. (2003) have considered the likely existence of critical windows during immune development when the immune system as a target may have increased sensitivity or increased resistance to xenobiotic-induced immunotoxic alteration. These windows correspond to different dynamic stages of functional development within the embryonic, fetal, and early neonatal immune system. One issue of immune development particularly pertinent to Pb is the fact that Th1 and Th2 functional capacities do not develop at the same time in either humans or rodents. The need to protect against maternal-fetal allogeneic reactions results in Th1 function being acquired largely after birth. Therefore, any environmental exposure that interferes with the rapid development of Th1 function might leave the individual with a Th2-biased immune system. The heavy metal Pb apparently represents one of the xenobiotics that are capable of suppressing Th1 capacity, resulting in dysregulated immune balance. This is discussed further in Sections 5.9.4, 5.9.8, and 5.9.10.

5.9.2 Host Resistance

Host resistance to disease has been used as an effective measure of the impact of environmental toxicants on immune function. Because different diseases require different combinations of immune effector functions for host protection, analysis of environmental modulation of host resistance across a spectrum of diseases can help identify clinically relevant immunotoxicity.

The 1986 AQCD presented a range of studies in which exposure to Pb inhibited host resistance to disease. Since the time of that report, few new infectious diseases have been added to the list of those that Pb is known to influence. Instead, a much broader understanding of the likely basis for the increased disease susceptibility to these pathogens has become evident. Additionally, recognition of an increased risk for some atopic and autoimmune diseases arising from lead-induced immunotoxicity has occurred in recent years. This is discussed under Section 5.9.8. Lead-induced alterations of host resistance against infectious and neoplastic diseases are considered in the following sections.

To date, there has been either no effect or an increased susceptibility to disease resulting from exposure to lead for virtually every infectious agent examined. Given the capacity of Pb to shift immune responses toward Th2, one might expect that enhanced resistance might occur for

diseases where robust Th2 responses were required. For example, an increased resistance against helminth parasitic disease might be hypothesized. However, this possible association has not been widely examined to date.

5.9.2.1 Viral Diseases

In general, exposure to Pb increases the susceptibility to viral infections. Studies include host resistance directed against the encephalomyocarditis virus (Gainer, 1977; Exon et al, 1979), Langat virus (Thind and Khan, 1978), and Semliki Forrest virus (Gupta et al., 2002). In the last example, oral dosing of Swiss mice with Pb-acetate (250 mg/kg for 28 days) significantly increased mortality to sublethal doses of the virus. Ewers et al. (1982) reported that occupational exposure to Pb resulted in an increased incidence of influenza cases among workers. In chickens administered Pb-acetate orally (20 and 40 mg/100g body weight) for 56 days, antibody production against Newcastle virus vaccine was reduced, while mortality against viral challenge was increased (Youssef et al., 1996). It seems likely that the reduced Th1 capacity (including effective CTL generation) combined with increased TNF-α, ROI, and prostaglandin E_2 (PGE_2) production by responding macrophages would contribute to increased tissue pathology but reduce viral clearance for many infections.

5.9.2.2 Bacterial Diseases

Most of the lead-associated host resistance research has been conducted on bacterial diseases. Hemphill et al. (1971) first described the increased susceptibility of mice exposed to Pb (250 µg given i.p. for 30 days) to *Samonella typhimurium*, while Selye et al. (1966) reported increased susceptibility of rats to bacteria endotoxins. Cook et al. (1975) found increased susceptibility of lead-exposed rats (2 mg/100g body weight given i.v. once) to both *Eschrichia coli* and *Staphylococcus epidermidis.*

The vast majority of studies have been conducted using the intracellular bacterium, *Listeria monocytogenes,* in mice. *Listeria* infection and host resistance to the disease have been well characterized. Essentially, this infection requires an effective antigen presentation (probably involving toll-like receptor 2 involvement), a robust response by activated macrophages leading to interlukin-12 (IL-12) and interferon-γ (IFN-γ) production and robust Th1 driven host protection (Torres et al., 2004; Lara-Tejero and Pamer, 2004). Ideally, activated

macrophages would produce NO in an effective response against *Listeria* (Ito et al., 2005). In the case of lead-induced immunotoxicity, everything works against this type of response. First, macrophages have severely suppressed NO production. Yet, overproduction of TNF-α, ROIs and PGE_2 leads to tissue inflammation and damage. The skewing of the response toward Th2 means that both IL-12 and IFN-γ are lacking. Excessive production of IL-6 and other pro-inflammatory cytokines results in what has been termed "sickness behavior" which involves both the immune and central nervous systems (Dantzer et al., 1998; Dyatlov et al., 1998a,b; Lawrence and Kim, 2000; Dyatlov and Lawrence, 2002). Lead-induced impairment in host resistance to *listeria* was reported by Lawrence (1981). CBA/J mice exposed orally to 80 ppm or greater of Pb-acetate for 4 weeks had 100% mortality (after 10 days) compared with no mortality for mice exposed to 0 or 16 ppm lead.

In an important study concerning individual variation to lead-induced immunotoxicity and host resistance, Kim and Lawrence (2000) demonstrated that neurological circuitry as it pertains to brain lateralized behavior could impact the effect of Pb on immune responses and host resistance to *Listeria*. Not surprisingly, this suggests that host genotype and epigenetic factors can be influenced by Pb exposure to the individual. Using female BALB/c mice, Kishikawa and Lawrence (1997) demonstrated that exogenously administered recombinant IL-12 (1 μg each for three days i.p.) could enhance production of IFN-γ as well as host resistance to *Listeria* in lead-exposed (2 mM in water for 3 weeks) mice. However, lead-exposed mice continued to have excess IL-6 production (part of the sickness behavior phenotype). The result with IL-12 validates the importance of the Th skewing and macrophage impairment induced by Pb on host resistance to certain diseases.

Additional bacterial infections in which Pb exposure has been reported to reduce host resistance include *Serratia marcesens* (Schlipkopter and Frieler, 1979) and *Pasteurella multocida* (Bouley et al., 1977).

5.9.2.3 Parasitic Diseases

Few studies have been conducted to date regarding the effects of Pb on host resistance to parasitic diseases. This is unfortunate as some parasitic disease challenges require effective Th2 responses for optimal resistance. Hence, it is not clear that Pb exposure would depress host resistance in every case (e.g., for helminth parasites). Since the AQCD in 1986, one study was

conducted examining the effect of Pb on the killing ability of *Leishmania enriettii* parasites in vitro by mouse macrophages (Mauel et al., 1989). The authors found that 30–100 mM Pb-acetate interfered with the killing ability of macrophages without producing macrophage cytotoxicity.

5.9.2.4 Tumors

The primary study concerning tumor immunity/tumor growth and Pb was already known at the time of the 1986 AQCD. In this study, male C57Bl/6 mice were exposed to Pb-acetate in the drinking water at concentrations of 0, 13, 130, or 1300 ppm. Moloney sarcoma virus (MSV)-induced tumor formation and growth were compared following the exposure of mice to Pb for 10-12 weeks. MSV-induced transplantable tumors were also used in this study. Primary tumor growth was enhanced in animals that received 130 and 1300 ppm of Pb vs. the control. However, all tumors regressed eventually. Most other studies involving Pb exposure and tumors describe the fact that Pb can exacerbate the ability of other toxins to promote tumor formation (Kobayashi and Okamoto, 1974; Hinton et al., 1979). Much of the tumor-promoting activity of Pb would seem to involve depressed Th1 and macrophage function as well as the promotion of excessive ROI release into tissues.

5.9.3 Humoral Immunity

The irony of Pb as an immunotoxicant is that the overall effects on humoral immunity are reasonably modest compared to those reported for macrophages and T lymphocytes (McCabe 1994). McCabe et al. (1991) discussed the fact that Pb is not profoundly cytotoxic for most immune cells yet can cause major functional shifts within the immune system as well as decreased host resistance to disease. In many cases, antibody production can remain robust in lead-exposed animals and humans. However, the nature and spectrum of the antibodies produced is the more significant cause for concern. Lead appears to alter the course of T lymphocyte-driven B cell maturation such that class switching may be skewed in lead-exposed animals and humans. If Pb dosage and duration of exposure is sufficient, antibody production may be depressed overall. However, with low-level Pb exposure, skewed isotype production is the greater health risk.

5.9.3.1 General Effects on B lymphocytes and Immunoglobulins

Despite the fact that T lymphocytes and macrophages appear to be the more sensitive targets of lead, the metal can alter B lymphocyte maturation and shift immunoglobulin production. The 1986 AQCD describes the fact that some early studies reported no effect of Pb on antibody production (Reigart and Graber, 1976; Ewers et al., 1982), while others reported a significant decrease in the humoral immune response (Koller, 1973; Koller and Kovaic, 1974; Blakley et al., 1980). In retrospect, this apparent discrepancy may have been caused by the various concentrations of Pb administered as well as variations in the duration of exposure. Additionally, as mentioned in the 1986 AQCD, the temporal relationship of Pb exposure to antigen challenge may be important.

In studies measuring generation of plaque forming cells (PFCs) against sheep red blood cells (SRBCs), Pb incubation with lymphocytes in vitro caused an increased response (Lawrence, 1981). In a comprehensive study using several strains of mice, Mudzinski et al. (1986) reported that Pb-acetate administered in the drinking water (10 mM for 8 weeks) elevated the response in the one strain (BALB/c mice) but failed to alter the humoral response to SRBCs (either PFCs or antibody titers) in all other strains. McCabe et al. (1990) reported that Pb caused an elevation in B cell expression of Class II molecules, thereby influencing B cell differentiation. Lead seemed to impact Class II molecule density at the cell surface via the levels of mRNA translational and/or the posttranslational stages of cell surface protein synthesis (McCabe et al., 1991).

Some human epidemiological and occupational studies have reported lead-associated differences in levels of circulating immunoglobulins. However, Tryphonas (2001) discussed the pitfalls of relying on total serum immunoglobulin in assessing immunotoxic effects in humans. Sun et al. (2003) reported that immunoglobulin M (IgM) and immunoglobulin G (IgG) were lower but that IgE was higher among females within their high-Pb group. Basaran and Undeger (2000) found that IgM, IgG, and some complement proteins were reduced among battery workers with high Pb exposure. Results of Undeger et al. (1996) were similar as well. In contrast, Sarasua et al. (2000) reported an elevation in immunoglobulin A (IgA), IgG, and IgM associated with environmental Pb exposure. Pinkerton et al. (1998) found no major effects but reported a significant lead-associated decline in serum IgG and an elevation in B cell percentage. In a human in vitro study, Borella and Giardino (1991) showed that Pb exposure caused an increased IgG production following stimulation of cells with pokeweed mitogen.

In more recent animal studies, Miller et al. (1998) and Chen et al. (1999) reported no effect on antigen-specific IgG titers against keyhole limpet hemocyanin (KLH) protein in F344 strain rats that had been exposed in utero to Pb (0–500 ppm Pb-acetate in drinking water).

It seems likely that Pb exposure may be capable of reducing serum immunoglobulin levels given sufficient dose and duration of exposure. However, the more critical issue pertains to the distribution of class and subclass of immunoglobulins produced after exposure to lead. Because Pb can alter the development of T cells involved in specific antigen responses, this can impact the spectrum of immunoglobulins produced in response to T-dependent antigens. As discussed in the following section, production of IgE (a class of immunoglobulin that is poorly represented in serum but of great clinical significance) is a central issue for lead-induced immunotoxicity. One additional health concern is the potential for Pb to enhance the likelihood of autoantibody production (Lawrence and McCabe, 2002; Hudson et al., 2003). This latter concern is discussed under Section 5.9.8.

5.9.3.2 IgE Alterations

One of the three predominant hallmarks of lead-induced immunotoxicity is an increase in IgE production. This can occur in the context of antigen-specific responses or as measured by total serum IgE. For this endpoint, the human and animal findings are very similar. Virtually all of the information concerning the capacity of Pb to elevate IgE production in humans and animals has been obtained since the 1986 AQCD. As a result, this represents a relatively new biomarker for lead-induced immunomodulation, and one not included in most animal or human studies conducted prior to 1990 (e.g., Wagerova et al., 1986).

Table 5-9.1 lists the studies reporting lead-induced elevation of IgE. The disease implications of lead-induced increases in IgE production are potentially significant and may help to address, in part, the allergy epidemic that has occurred in the last several decades (Isolauri et al., 2004). A relationship has been established between relative Th2 cytokine levels, serum IgE levels, and the risk of allergic airway inflammation (Maezawa et al., 2004; Cardinale et al., 2005). In fact, attempts to manage allergic inflammation use IgE as one of the major targets (Stokes and Casale, 2005). IgE levels are directly related to the production of Th2 cytokines such as interlukin-4 (IL-4), among others (Tepper et al., 1990; Burstein et al., 1991; Carballido et al., 1995; Takeno et al., 2004; Wood et al., 2004). The relationship between Th2 cytokines

Table 5-9.1. Recent Studies Reporting Lead-Induced Increase in IgE

Species	Strain/Gender	Age	In vivo Ex vivo	Lowest Effective Dose	Exposure Duration	Reference
Human	Both genders	Children	Yes	Not available	Not Available	Karmous et al. (2005)
Human	Both genders, 91% males	Adult	Yes	Not Available	Not Available	Heo et al. (2004)
Human	Females	Children	Yes	Not Available	Not Available	Sun et al. (2003)
Mouse	Balb/c males and females	Fetal	Yes	0.1 mM	3 days	Snyder et al. (2000)
Human	Both genders, 56% male	Juvenile	Yes	Not Available	Not Available	Lutz et al. (1999)
Rat	F344 females	Embryo – fetal	Yes	100 ppm	5 weeks to dam (2 and 3 gestational)	Miller et al. (1998)
Mouse	Balb/c females	Adult	Yes	50 µg 3x per week s.c.	3 weeks	Heo et al. (1996)
Human	Males	Adult	Yes	Not Available	Not Available	Horiguchi et al. (1992)

(e.g., IL-4), IgE levels, and allergic airway disease is supported through various pharmacological interventions in both animals and humans that either induce Th2 cytokine and promote allergic airway disease (Wu et al., 2004) or interfere with Th2 cytokine-driven IgE production and inhibit allergic inflammation (Holgate et al., 2005; Ban and Hettich, 2005). The production of IgE is of importance in terms of potential inflammation. Not only is the level of IgE a consideration, but also the expression of the Fc receptor for the epsilon (ε) chain of IgE on mast cells and basophils.

In humans, Karmaus et al. (2005) reported a positive association of blood Pb levels with serum IgE concentration among second grade children living near a waste incinerator or other lead-emitting industries. Sun et al. (2003) also found a positive association of blood lead and serum IgE levels among children in Taiwan. Lutz et al. (1999) reported a correlation of blood lead levels and serum IgE levels in children in Missouri from 9 months–6 years of age. This association appears to hold not only for children but also for adults. Heo et al. (2004) recently showed that battery workers with blood leads > 30 µg/dL differed significantly in serum IgE levels from those with blood leads < 30 µg/dL. Additionally, serum IgE concentration correlated with blood lead among the populations examined (r = 0.0872).

Animal data support this relationship between blood lead concentration and IgE level and further suggest that even very low-level Pb exposure early in development may produce elevated IgE production in the juvenile offspring. Miller et al. (1998) found that gestational exposure of rats to 100 ppm Pb-acetate in the drinking water could produce elevated IgE in the adult offspring. Snyder et al. (2000) showed that gestational and/or neonatal exposure of mice to Pb-acetate produced neonatal blood lead levels not above background (5.0 μg/dL), but nevertheless, could result in elevated IgE production in the juvenile mouse. In most cases, Pb exposures associated with elevated IgE were also associated with increases in IL-4 production by T cells (Chen et al., 1999; Snyder et al., 2000). This is consistent with the fact that high IL-4 production can predispose B lymphocytes to undergo a specific class switch for the production of IgE.

One importance of these findings is that, in each case, the elevation in IgE persisted long after blood Pb levels had returned to normal. This means that Pb exposure could occur early in life and produce an increased risk of later-life allergic disease with no residual evidence that the individual had ever been exposed to lead. This should provide a cautionary note for future human studies examining Pb body burden and immune function.

5.9.4 Cell-Mediated Immunity

Cell-mediated immunity (CMI) essentially involves all host resistance beyond the soluble components of defense, i.e., antibody and complement. CMI includes any action of the immune system that is a direct effect of leukocytes on neoplastic or virally-infected cells or against extracellular targets such as bacteria. Even macrophage functional processes involving antibodies, such as antibody-dependent cellular cytotoxicity (ADCC), are considered to be CMI. One of the hallmarks of CMI is that cellular activation is usually required for the effector cells to attack the target. In the case of macrophages, this is usually activation with the Th1-associated cytokine, IFN-γ.

For NK cells, activation can occur through various pathogenic components such as double stranded RNA. However, recently Borg et al. (2004) showed that mature dendritic cells produced a Th1-promoting cytokine, IL-12, and this in turn activates NK cells to produce the further Th1-promoting cytokine, IFN-γ. Interleukin-18 (IL-18) produced by macrophages is also an activator of NK cells, facilitating Th1-promoting cytokine release while interleukins-2 and -15 (IL-2, IL-15) are growth factors for NK cells. NK cells would appear to be relatively

resistant to the effects of Pb compared to some T lymphocytes and macrophages. For a detailed consideration of the effects of Pb on NK cells, see Section 5.9.7.

Cytotoxic T lymphocytes are generated in response to antigen presentation delivered with Th1 cytokines. These cells are capable of mediating antigen specific destruction of neoplastic and virally-infected cells via binding and release of cytolytic proteins into the intracellular space. Frequently, the most effective antigen targets of CTLs are the early viral proteins produced in the first phase of host cell infection by viruses. IL-12, produced largely by dendritic cells, appears to be important in the generation of antigen CTL cells and IFN-γ produced by Th1 lymphocytes. NK cells are a potent regulator of CTL activity. Cell signaling via certain toll-like receptors on antigen presenting cells seems to have a role in determining the nature of the Th activation (Th1 vs. Th2) and can, therefore, influence the extent of CTL production.

Because T lymphocytes and their regulator and effector functions are so critical in CMI, the maturation of thymocytes within the thymus microenvironment and the selection of repertoire among the maturing T lymphocytes are crucial issues for potential developmental immunotoxicants. In fact, lead seems to be capable of disrupting several aspects of T cell maturation, activation, and repertoire usage (McCabe and Lawrence, 1991; Heo et al., 1998; Miller et al., 1998; McCabe et al., 2001, Lee and Dietert, 2003).

5.9.4.1 General Effects on Thymocytes and T lymphocytes

In general, cells of the T cell lineage appear to be relatively sensitive to the toxic effects of Pb compared to other lymphoid populations. At the time of the 1986 AQCD, there was some understanding of this sensitivity. However, there appear to be considerable differences in sensitivity across various T cell subpopulations (McCabe and Lawrence, 1991; Heo et al., 1996; 1997; 1998). This was largely unknown when the prior AQCD was prepared as the partitioning of T helper cells into functionally distinct subpopulations (e.g., Th0, Th1, and Th2) was not known until the latter part of the 1980s. The differential impact of Pb on T helper cell populations and on immune balance was established during the 1990s. This has become one of the four hallmarks of lead-induced immunotoxicity.

Original observations of both in vivo and in vitro T-dependent immune responses in the presence of Pb suggest that T helper function, as well as the spectrum of cytokines produced, are skewed toward the Th2. The cytokine skewing is discussed as well in Section 5.9.5.3. Smith

and Lawrence (1988) have shown that Pb can inhibit antigen presentation and stimulation of a T cell clone of the Th1 phenotype. McCabe and Lawrence (1991) were the first to show that this was caused by the novel capacity of Pb to inhibit Th1 stimulation while promoting presentation to Th2 clones. Heo et al. (1996) provided both in vitro and in vivo results supporting this immunomodulation of lead. Cytokine skewing accompanied the differential stimulation of Th cells.

Using naïve splenic CD4+ T cells derived from D11.10 ovalbumin-transgenic mice, Heo et al. (1998) developed T cell clones in vitro in the presence of lead. The authors found the T cells that developed from the naïve precursors were significantly skewed toward the Th2 helper phenotype and away from the Th1 phenotype. If IL-4 was inhibited with the addition of anti-IL-4 to the cultures or if the Th1- promoting cytokine IL-12 was added exogenously to the culture, the effects of Pb could be largely overcome. This study provided firm evidence that Pb can directly promote Th2 development among precursor Th(0) cells and impair development of Th1 cells. Among its effects, Pb enhanced adenyl cyclase activity and increased the levels of cAMP. The authors suggested that Pb may influence cell signaling in such as manner as to promote the Th2 pathway.

Beyond the biasing of immune responses at the level of the T lymphocyte based on Th1/Th2 balance, Pb has the capacity to bias usage of certain Vβ genes (Vβ 5, Vβ 7, and Vβ 13) among T lymphocyte clones in mice (Heo et al., 1997). This is of concern, as it suggests that exposure to Pb may alter the T cell repertoire and skew its representation. Heo et al. (1997) discussed the fact that many autoimmune diseases are characterized by a disproportionate usage of certain Vβ genes. Different autoimmune conditions are associated with the differential overabundant usage of a specific Vβ gene. They suggest that this feature of lead-induced T lymphocyte immunotoxicity may contribute to and enhance the risk of autoimmunity.

Lee and Dietert (2003) exposed the developing thymus of embryonic day 12 (E12) chickens to Pb-acetate (single injection of 400 μg) and evaluated the capacity of thymocytes (ex vivo) from juvenile chickens to produce IFN-γ. They found that embryonic exposure at doses that impair juvenile delayed type hypersensitivity (DTH) also inhibit IFN-γ production. Similarly, IFN-γ production was decreased when thymocytes from juvenile chickens were exposed to Pb in vitro (0.45 μM). However, in vitro exposure of thymic stroma to Pb did not result in suppression of control thymocyte IFN-γ production in co-cultures. There is a suggestion

that the balance of reproductive hormones in early life may influence the impact of Pb on developing thymocytes (Hussain et al., 2005).

5.9.4.2 Delayed Type Hypersensitivity

The DTH assay is an in vivo assay requiring antigen-specific T lymphocytes to be primed, expanded, and then recruited to a local site of antigen deposition. The most common application of the DTH is the tuberculin assay for TB in humans. The assay has a long history of application in immunotoxicology, and its utility within the national toxicology program assessment in the mouse has been previously reported (Luster et al., 1992). The assay is known to depend largely on Th1 participation and is, therefore, an effective measure of Th1-dependent function. However, there are at least two different portions of the response that are under somewhat separate control. Priming and expansion of the antigen-specific T lymphocytes is largely Th1 dependent. However, recruitment of T lymphocytes to the periphery involves a variety of locally-produced chemotactic signals that may not be under the same regulation. In fact, Chen et al. (1999) showed that a commonly used chelator for Pb poisoning (succimer, meso-2,3-dimercaptosuccinic acid [DMSA]) fails to restore lead-induced suppression of DTH in rats, because the chelator itself somehow interferes with the production of chemotactic factors necessary for T lymphocyte recruitment. The DTH assay is also generally useful in questions of possible developmental immunotoxicity, because of the natural skewing toward Th2 that occurs during gestation through birth and the issue of effective Th1 functional acquisition in the newborn.

Lead-induced suppression of the DTH response is one of the four hallmarks of lead-induced immunotoxicity. At the time of the 1986 AQCD, the capacity of Pb to suppress DTH function was already known from two studies conducted during the late 1970s. However, the association of the function with Th1 help had not been established. Muller et al. (1977) were among the first to demonstrate lead-induced suppression of DTH. Using mice, these investigators administered Pb-acetate i.p. for 30 days prior to assessment of primary and secondary DTH responses against SRBCs. Both primary and secondary responses were severely depressed following exposure to Pb, even at the lowest dose tested (0.025 mg). Faith et al. (1979) exposed developing Sprague-Dawley rats to Pb-acetate in the drinking water (lowest dose at 25 ppm) first via the dams during gestation and through weaning and then with direct exposure

of the offspring until 6 weeks of age. In this case, the purified protein derivative (PPD) of tuberculin was used as the antigen compared against the saline injection control. Rats administered the lowest dose of Pb evaluated (producing a BLL of 29.3 µg/dL) had a significantly reduced DTH response. Laschi-Loquerie et al. (1984) measured the contact hypersensitivity reaction against picryl chloride in mice that had received 0.5 mg/Kg Pb via s.c. administration. Lead administration was given from 3-6 days in duration at varying times relative to the sensitization period. These investigators reported that Pb suppressed the DTH type of response regardless of the window (before or during sensitization) in which it had been administered.

More recently, Miller et al. (1998) found that female F344 rats gestationally exposed to 250 ppm of Pb-acetate in drinking water had a persistently reduced DTH reaction against KLH protein. Chen et al. (1999), Bunn et al. (2001a,b,c) and Chen et al. (2004) had similar findings in studies that included both the F344 and CD strains of rats. In the last study conducted in F344 rats, a BLL of 6.75 µg/dL at 4 weeks of age, postgestational exposure to Pb-acetate (250 ppm in drinking water) was associated with depressed DTH against KLH in the 13-week-old adult female offspring (Chen et al., 2004). McCabe et al. (1999) were among the first to draw attention to the relationship between lead-induced suppression of DTH and the prior observations of lead-induced Th skewing. These authors gave varying doses of Pb-acetate in drinking water (32,128, 512, 2048 ppm) to female BALB/c mice for 3 weeks prior to measuring the DTH against SRBCs. They found that the 512 ppm dose producing a BLL of 87 µg/dL significantly impaired the DTH response. Antigen routes proved to be important as Pb depressed DTH when an i.v. primed with SRBCs was used, but not when SRBCs were administered i.p. Timing of Pb administration was found to be important relative to the capacity to depress the DTH response. Lee et al. (2001) showed that Pb-acetate (200 µg) administered in ovo to chicken embryos at 9 days of incubation failed to depress juvenile DTH against bovine serum albumin (BSA), but when the same dose of Pb was administered 3 days later producing the same BLL, juvenile DTH was severely reduced. Using the latter model, embryonic administration of exogenous thymulin was found to partially restore juvenile DTH function following embryonic exposure to Pb (Lee and Dietert, 2003).

Regarding developmental sensitivity of the DTH response to lead-induced immunosuppression, parallel findings were obtained in the developing rat (CD strain females)

(Bunn et al., 2001c) in agreement with those found in the chicken. Administration of 500 ppm Pb-acetate during gestational days 3 to 9 or 15 to 21 produced no DTH effect compared with DTH suppression in the corresponding adult offspring. As shown in Figure 5-9.1, the sensitivity of the DTH response to Pb appears to develop sometime between days 9 and 15 of rat embryonic development. Apparently, the status of the developing thymus may be a consideration in the capacity of Pb to impact the subsequent DTH response. This is discussed further in Section 5.9.10.

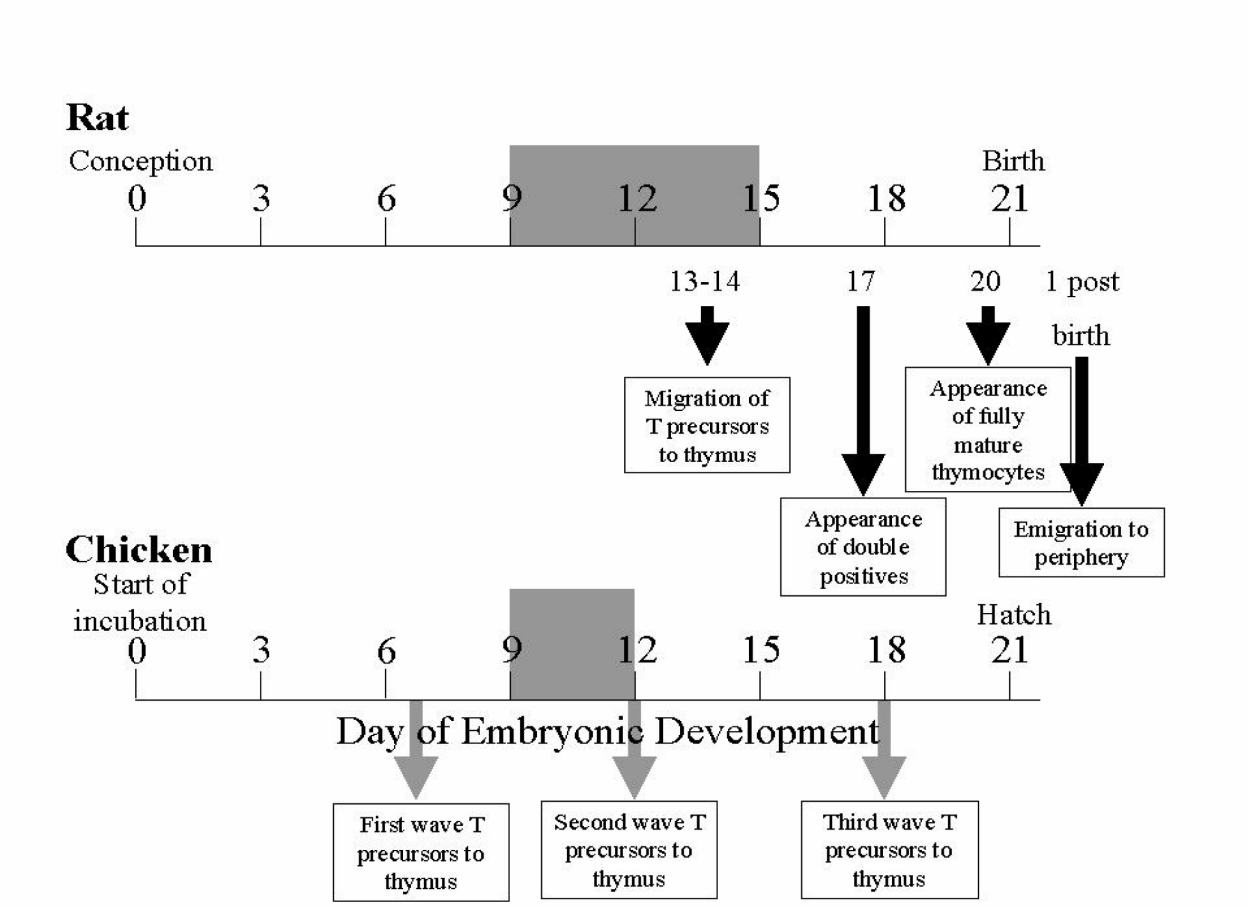

Figure 5-9.1. Windows during prenatal development (days postconception for rat) or embryonic development (days postincubation initiation for chicken) during which sensitivity of DTH to lead emerges.

It should be noted that in several studies, lead-induced suppression of the DTH response was associated with reduced capacity to produce the Th1 cytokine, IFN-γ (Chen et al., 1999; Lee et al., 2001).

5.9.4.3 Other T-Dependent Cell-Mediated Immune Changes

The in vitro response of T lymphocyte populations to various mitogens (e.g., Concavavalin A [ConA], Phytohemagglutinin A [PHA]) has been used as a surrogate measure of antigen-driven T lymphocyte stimulation. The impact of Pb on these parameters is presented in Section 5.9.5. Another T cell response altered by exposure to Pb is the mixed lymphocyte response (MLR). This in vitro assay is a measure for the responsiveness of T cells to the presentation of allogeneic major histocompatibility complex (MHC) molecules by antigen presenting cells. The in vivo correlate of the MLR is usually considered to be the graft vs. host (GvH) reaction. Several investigators have reported Pb alteration of the MLR as summarized in Table AX5-9.4.

McCabe et al. (2001) demonstrated that Pb at very low physiological concentrations (0.1 μM or approximately the equivalent of 10 μg/dL) in vitro significantly enhanced the proliferation and expansion of murine alloreactive CD4+ T lymphocytes in the MLR reaction. In fact, the resulting population was found to have a high density of CD4 molecules on the cell surface-making them phenotypically similar to memory T lymphocytes. The authors hypothesized that lead-induced creation of an exaggerated pool of memory-type T lymphocytes (possessing a lower threshold required for subsequent activation) would be problematic for the host. In a study using Lewis strain rats, Razani-Boroujerdi et al. (1999) also found evidence for lead-induced stimulation of the in vitro MLR response. In this case, both the alloreactive mixtures of cells as well as syngeneic mixtures were elevated in proliferation when cultured in the presence of Pb-acetate (e.g., 50 ppm or approximately 131 μM). When concentrations of Pb were significantly higher (200 ppm or greater), proliferation was inhibited in these cultures.

Figure 5-9.1 illustrates the developmental appearance of initial sensitivity for Pb-induced suppression of the DTH function. The mid-embryonic developmental window is the time during which the capacity of Pb to impair later-life DTH responses first emerges. Earlier pulsed exposure to Pb fails to impair juvenile and/or adult DTH despite the continuing of presence of Pb in the embryo. However, during the second half of embryonic development the embryo becomes remarkably sensitive to lead-induced suppression of DTH. Both the rat and chicken are similar in this window of emerging Th1-dependent functional sensitivity. Thymus-related developmental events are indicated along with the emergence of DTH functional sensitivity to

lead. Information was derived from Gobel (1996), Vicente et al. (1998), Dunon et al. (1999), Dietert et al., (2000), Bunn et al. (2001c), Lee et al. (2001) and Holsapple et al. (2003).

5.9.5 Lymphocyte Activation and Responses

Many of the broader functional ramifications of Pb exposure on lymphocytes are discussed under Sections 5-9.3 and 5-9.4. However, the capacity of Pb to directly alter lymphoid responses is a significant component of lead-induced immunotoxicity and is summarized within the present section. Lymphoid responses are usually assessed in terms of proliferation and activation (functional changes). One of the recent endpoints reflecting functional status is the production of cytokines. These both autoregulate the producing cells and significantly impact the activity of other immune and nonimmune cells carrying the appropriate receptors. The spectrum and levels of cytokines produced by a population of immune cells tends to reflect their capacity to regulate the host immune response.

5.9.5.1 Activation by Mitogens

The capacity of certain plant- and bacterially derived products to stimulate lymphoid populations to enter the cell cycle and undergo mitogenesis has been used for decades to assess the potential capacity of lymphocytes to receive proliferation signals and expand their population. Among the mitogens employed within the Pb exposure studies are the T lymphocyte subpopulation mitogens, PHA and Con A; the dual T and B cell mitogen, pokeweed mitogen (PWM; the B lymphocyte mitogen derived from gram-negative bacteria, lipopolysaccharide (LPS), and the B cell mitogen, *Staphylococcus aureus* enterotoxin (SE). It should be noted that these mitogens do not necessarily stimulate all T lymphocytes or all lymphocytes but, instead, stimulate selected populations of the cells. The mitogens react with a large array of cell surface molecules producing cross-linking and appropriate signal transduction to initiate mitogenesis. In the case of the plant-derived mitogens, lectins, numerous glycoproteins and glycolipids carrying the correct carbohydrate residues serve as the cell surface binding sites for cross-linking. Mitogen stimulation in vitro has been used as a surrogate for antigen-driven stimulation and proliferation of antigen-specific T and B cell clones. However, it should be noted that while the assays have been used for decades, there are now more specific assays utilizing more functionally relevant cell surface receptors to assess lymphoid activation potential.

The 1986 AQCD has an extensive review of mitogenic responses of lymphocytes following both in vivo and in vitro treatment by lead. The results at that time showed no clear pattern. At low to moderate levels, Pb was potentially co-mitogenic for some cells and at very high concentrations could suppress proliferation. Little has changed in conclusions for this assessment measure since the 1986 report. The most significant findings from the mitogenic studies are that at doses encountered physiologically Pb is not a potent cytotoxic agent for most immune cells. At low concentrations, it can marginally stimulate lymphoid mitogenesis. However, as one examines more refined subpopulations of lymphocytes than what were able to be identified prior to 1986 (e.g., Th1 vs. Th2 clone of T lymphocytes), it becomes clear that Pb can promote expansion of some lymphoid populations while suppressing others.

Annex Table AX5-9.5 for this section summarizes results of Pb effects on mitogen-stimulated proliferation of lymphoid populations.

5.9.5.2 Activation via Other Receptors

In recent years, lymphoid activation and population expansion has been measured using the triggering of specific T and B cell surface receptors (e.g., CD3 on T cells) as well as antigen-driven proliferation of T cell clones known to be specific for the antigen in question. The latter has provided the opportunity to simulate in vivo lymphoid activation and antigen-driven proliferation by using receptors in vitro, which are more physiologically relevant than those activated by plant lectins. Because Pb does not cause profound population loss across the entire population of T or B lymphocytes, these more refined and functionally-relevant assay systems have enabled a much clearer picture to emerge concerning lead-induced changes in lymphoid population than was available for the 1986 AQCD report.

Smith and Lawrence (1988) and McCabe and Lawrence (1991) utilized antigen-specific mouse T clones. They found that Pb directly promoted antigen presentation and stimulation of the T cell clones when these clones were Th2 cells. However, when the Th1 clones were used, Pb suppressed the antigen-specific presentation signal. In the McCabe and Lawrence study, direct comparisons were made between Th1 and Th2 clones specific for mouse allogeneic MHC molecules. These studies provided the first clear picture of the differential effects of Pb on Th1 vs. Th2 cells. Several studies since these have verified this major effect of Pb (Heo et al., 1996; 1997, 1998). Many of these later studies utilized the transgenic mouse strain (DO11.10 OVA-tg)

that carries T cells specific for a peptide fragment of ovalbumin. These enabled the same comparisons to be made with the presentation of a soluble protein antigen as the stimulating signal. Heo et al. (1998) showed that Pb not only selectively stimulates Th2 cells and suppresses Th1 cells but that it preferentially causes precursor Th0 cells to mature into Th2, rather Th1 cells, as well. Additionally, the T cell clones in the presence of Pb are skewed in terms of their usage of Vβ genes (as reflected in their cell surface receptors) (Heo et al., 1997). This is of particular concern relative to the risk of autoimmunity. More recently, McCabe et al. (2001) examined Pb exposure in the context of the allogeneic MLR against allogeneic MHC molecules. In vitro exposure to Pb (as low as 1.0 μM) enhanced the primary MLR response, but not the secondary MLR response and not the mitogenic response using PHA. Significantly, the T cell clones that emerged from the primary MLR were in greater proportion than normal and were of the specialized phenotype CD4-plus high density ($CD4+^{high}$). Because these fit the phenotype of memory cells, it is likely that an overabundance of memory cells was produced during the primary response, where the antigen may be of lesser biological significance than in a secondary response. The authors discussed the fact that Pb may cause T cells to respond under conditions of low antigen concentration, which could waste valuable and limited resources by generating T memory cell clones when they are not needed (against unimportant antigens) or even increase the risk of autoimmune responses by altering the threshold requirements for stimulation. The putative mechanisms suggested for the differential effects of Pb on Th cells are presented in Section 5.9.9.

5.9.5.3 Cytokine Production

At the time of the 1986 AQCD, immune cytokines were essentially absent from the information available for consideration. Only the antiviral interferons (α,β) had been examined among studies available for that report. Therefore, one of the most important effects of Pb on the immune system, i.e., Pb-induced cytokine production was not known at that time.

Most studies since 1986 have shown that Pb exposure at low to moderate levels causes a significant shift in the production of Th1 vs. Th2 cytokines with the bias toward the latter. Hence, production of IFN is decreased and IL-12 is inadequate for effective host resistance. In contrast, production of IL-4, IL-6, and, frequently, interlukin-10 (IL-10) is elevated. Table 5-9.2 illustrates the studies reporting shifts in cytokine production induced by lead.

Table 5-9.2. Studies Reporting Lead-Induced Shifts in Th1 vs. Th2 Cytokines

Species	Strain/ Gender	Age	Cytokine Alterations	In vivo/ Ex vivo	Lead Dose/ Concentration	Duration of Exposure	References
Rat	F344 Females	Embryo-fetal	↑IL-4 ↓IFN-γ splenic lymphocytes	Yes	250 ppm in water to dams	2 weeks prior and 3rd week of gestation for dam	Chen et al. (2004)
Human	Males	Adults	↑ IFN-γ PHA stimulated peripheral blood lymphocytes	Yes	Not available	Not available	Mishra et al. (2003)
Chicken	Cornell K females	Embryonic	↓IFN-γ stimulated thymocytes	Yes	400 μg	Single injection E12	Lee and Dietert (2003)
Mouse	Balb/c	Neonatal/ Juvenile	↑IL-6 serum during infection	Yes	0.5 mM in water to dams and their pups	4 weeks (3 via dams)	Dyatlov and Lawrence (2002)
Rat	CD females	Fetal	↑IL-10	Yes	550ppm in water to dams	6 days via gestation of dam	Bunn et al. (2001c)
Chicken	Cornell K females	Embryonic	↓IFN-γ	Yes	50 μg	Single injection	Lee et al. (2001)
Mouse	Balb/c male	Adults	↑IL-6 serum during infection in certain groups	Yes	2 mM	8 weeks	Kim and Lawrence (2000)
Mouse	NOD Autoimmune strain adult	Adult	↓IFN-γ, no change long term ↓TGF-β intestinal levels	Yes	Oral 10 mM and ovalbumin antigen	10 days	Goebel et al. (2000)
Mouse	C57 Bl/6 females NOD autoimmune strain females	Adult	No effect on gut balance in normal mice ↓TGF-β in autoimmune mice	Yes	0.5 mg/kg injection and oral ovalbumin	6 injections over 2 weeks	Goebel et al. (1999)
Rat	F344 females	Embryo-fetal	↓IFN-γ ↑IL-10	Yes	250 ppm to dams	2 weeks before and 3rd week of gestation	Chen et al. (1999)

Table 5-9.2 (cont'd). Studies Reporting Lead-Induced Shifts in Th1 vs. Th2 Cytokines

Species	Strain/ Gender	Age	Cytokine Alterations	In vivo/ Ex vivo	Lead Dose/ Concentration	Duration of Exposure	References
Mouse	DO11.10 ova-tg, ova mice and RAG knockouts	Adult	↓IFN-γ	No	25 μM	3 days	Heo et al. (1998)
Rat	F344 females	Embryo-fetal	↓IFN-γ	Yes	500 ppm to dams	2 weeks before and 3rd week of gestation	Miller et al. (1998)
Mouse	Balb/c ByJ females	Adult	↓IFN-γ ↑IL-6	Yes	2 mM	3 weeks	Kishikawa et al. (1997)
Mouse	Balb/c and DO11.10 ova-tg mice	Adult	↓IFN-γ ↓IFN-γ/IL-4 ratio	Yes	50 μg each injection (s.c.) 3 per week	2 weeks	Heo et al. (1997)
Mouse	Balb/c ByJ female or male	Adult	↓IFN-γ ↑IL-4	Yes	50 μg each injection (s.c.) 3 per week	2 weeks	Heo et al. (1996)
Mouse	Balb/c ByJ female or male	Adult	↓IFN-γ ↑IL-4	No	10 μM – 50 μM	2 days	Heo et al. (1996)

(Please note that TNF-α production is considered in the macrophage section, Section 5.9.6). These shifts in cytokine production are remarkably consistent, occur even at low levels of exposure, and are reported following both in vivo and in vitro exposure to lead. Furthermore, the effects are persistent even when exposure to Pb was restricted to early development and cytokine assessment was performed in the subsequent juvenile or adult (Miller et al., 1998; Bunn et al., 2001c; Lee et al., 2001; Chen et al., 2004).

The only exceptions to lead-induced biasing in favor of Th2 occur in the reports by Goebel et al. (2000) and Mishra et al. (2003). In the latter case, the authors attributed this difference (in humans) to the very high Pb levels considered in the study. In the prior case, Goebel et al. (2000) saw a local bias to Th1 in the intestinal tract of a specialized autoimmune diabetes-prone strain of mice (NOD) but not in normal mice. Initially, the Pb-induced cytokine skewing favored Th2 (after 1 day), but this shifted to Th1 with more prolonged Pb exposure (after 10 days). Loss of oral tolerance accompanied this long-term shift. These results suggest that in most cases, lead-induced skewing would favor Th2. But with some genotypes or additional disease conditions, an imbalance may occur in the direction of a gut-associated Th1 environment, increasing risk for loss of oral tolerance and the potential for increased food allergies.

One ramification for the capacity of Pb to promote Th2 cells is the impact of elevated IL-4 on IgE. It seems clear that lead-induced overproduction of IgE (seen in virtually all animal models examined as well as humans) is directly linked with the overproduction of IL-4. Excessive IL-4 and the resulting IgE production increases the risk for IgE-mediated atopy and asthma.

Additionally, Kishikawa et al (1997) demonstrated that administration of the potent Th1-promoting cytokine, IL-12, to lead-exposed mice can restore the balance of Th1 (IFN-γ) vs. Th2 cytokines (e.g., IL-6), reduce corticosterone levels, and enhance host resistance in *Listeria*-infected mice. This observation supports the critical role of Th1/Th2 balance in overall risk to host resistance against disease presented by Pb disruption of that balance.

5.9.6 Macrophage Function

Macrophages represent a diverse population of cells that play critical roles in both host defense and tissue homeostasis. Macrophage subpopulations provide a front line of defense

against bacteria, parasites, viruses, and tumors via the innate immune response. Additionally, they are important in tissue repair and remodeling as well as in the removal of senescent cells. Some forms of macrophages are efficient in the processing of antigens and the presentation of antigen fragments to T lymphocytes. Additionally, macrophages can regulate lymphoid activity through the secretion of a variety of cytokines and through the production of various immunomodulatory metabolites (e.g., NO, ROIs) and the products of the cyclooxygenase and lipoxygenase pathways.

Because macrophages can be found residing in most tissues, lead-induced modulation of macrophage functional capacity has the potential to alter overall organ function. Macrophages originate in the bone marrow from pluripotent stem cells that give rise to both the monocyte-macrophage lineage as well as polymorphonuclear leukocyte populations. Bone marrow-derived macrophages mature under the influence of various cytokine growth factors to become the full array of mature cell subpopulations. Various investigators have examined the effects of lead on the maturation of macrophages in vitro as well as on the functional capacity on fully mature cells both in vitro and in vivo. Blood monocytes represent a functional, yet not fully specialized, form of macrophage. As a result the influence of environmental toxicants on monocytes may not be fully predictive of the effects of the same toxicants on splenic or alveolar macrophages, glial cells, or Kupffer cells.

Because macrophages give rise to several specialized populations, e.g. Kupffer cells in the liver, glial cells in the brain, and various skin macrophage populations, it is important to realize that different specialized macrophage populations are likely to have somewhat different sensitivities to lead, as well as potentially different responses following exposure. Not surprisingly, blood monocytes may not always be an appropriate model to accurately predict the outcome of lead-induced immunotoxicity for alveolar macrophages following an inhalation exposure.

The 1986 AQCD identified macrophages as a significant target for lead-induced immunotoxicity. Research since the mid-1980s has served to underscore this point. The understanding of lead-induced alterations in macrophage function has increased significantly since the prior AQCD report. The following sections describe the reported immunotoxic effects of lead on macrophages. It should be noted that for a number of endpoints, such as lead-induced alterations in the production of NO, ROIs and TNF-α, there is a general consensus among a

majority of immunotoxicology studies and agreement with the effects described for the cardiovascular system (see Chapter 5.5).

5.9.6.1 Nitric Oxide (NO) Production

Nitric oxide is a short-lived metabolite produced in large quantities by macrophages during cellular activation. The enzyme responsible is an inducible form of nitric oxide synthase (iNOS), which, utilizing a bioptrin cofactor, converts the amino acid arginine into NO and citrulline. A competing alternative pathway utilizing arginine leads to the production of polyamines, which themselves are immunomodulatory for lymphocytes. Nitric oxide is critical in the defense against certain infectious agents, including various bacteria.

Among the most sensitive immunomodulatory effects of Pb exposure is the capacity to impair NO production by macrophages (Table AX5-9.6). Several research groups have shown that in vitro as well as in vivo exposure to Pb results in significantly reduced production of NO (Tian and Lawrence, 1995, 1996; Chen et al., 1997; Lee et al., 2001; Pineda-Zavaleta et al., 2004 [also reviewed in Singh et al., 2003]). Similar results were obtained in human, mouse, rat and chicken. Depression of NO production capacity usually occurs shortly after exposure to lead. However, the long-term effects of Pb on NO production following very early life exposure are less clear (Miller et al., 1998; Chen et al., 1999; Bunn et al., 2001a).

Tian and Lawrence (1996) have hypothesized that because very low Pb concentrations (in vitro equivalents to 10 μg/dL) can impair NO production, impaired NO production may be responsible for reduced host resistance to *Listeria* seen among lead-exposed rodents as well as for lead-induced hypertension among humans (Pirkle et al., 1985). Indeed, impaired NO production by macrophages seems to be one of the more sensitive endpoints for immediate lead-induced immunotoxicity.

5.9.6.2 Other Functional Alterations

TNF-α Production

Early studies identified the fact that Pb exposure could predispose animals for a dramatically increased sensitivity to bacterially-derived endotoxin (Trejo et al., 1972; Filkins and Buchanan, 1973; Schlick and Friedberg, 1981).

It is now known that the increased sensitivity to endotoxin is linked to the capacity of Pb to increase production of TNF-α among macrophages (Dentener et al., 1989; Zelikoff et al., 1993; Guo et al., 1996; Miller et al., 1998; Chen et al., 1999; Krocova et al., 2000; Flohe et al., 2002). Studies in mouse, rat, rabbit, and human provide a clear indication that one effect of Pb on macrophages is to boost production of the proinflammatory cytokine TNF-α. While most studies examined the immediate effects of Pb exposure on TNF-α production, studies by Miller et al. (1998) and Chen et al. (1999, (2004) showed that the effects of early gestational exposure to Pb on macrophages could persist well into later life, including adulthood. Additionally, Chen et al. (1999) showed that chelation of Pb with succimer in developing female rats in utero could eliminate the persistent effect of elevated TNF-α production in the adult offspring. Flohe et al. (2002) found evidence that lead-induced elevation in TNF-α production is sensitive to both PKC signaling as well as to protein production. While the production of TNF-α can be elevated following exposure to lead, the expression of the receptor for TNF-α (TNF-R) was also increased during the in vitro exposure of human blood monocytes to Pb-chloride (Guo et al., 1996). Therefore, the combined effect of elevated cytokine production by macrophages as well as increased receptor expression would be expected to contribute to problematic inflammatory responses.

Production of Other Proinflammatory Cytokines

Several studies have indicated that macrophage production of cytokines (or that levels of cytokines known to be produced primarily by macrophage populations) is altered after exposure to lead. These vary somewhat, depending upon the exposure protocol and the source of macrophages examined. In addition to the previously discussed elevation of TNF-α by lead, the most significant and consistent lead-induced effects seem to involve elevated production of the other major proinflammatory cytokines, interleukin-1β (IL-1β) and IL-6. Increased production of IL-6 following exposure to Pb has been reported by Dyatlov and Lawrence (2002), Flohe et al. (2002), Kim and Lawrence (2000), Krocova et al. (2000), Kishikawa and Lawrence (1998) and Kishikawa et al. (1997). Because IL-6 is a proinflammatory cytokine, its increased production following Pb exposure has the potential to influence many different tissues. Dyatlov et al. (1998a,b) provided evidence that lead, IL-6 and LPS can combine to exert a significant impact on the permeability of the blood brain barrier as well as the properties of brain neurons and

endothelial cells. Lead-induced elevation of IL-1β production has been reported by Dyatlov and Lawrence (2002). It is probable that enhanced co-production of IL-1β and IL-6 would increase the likelihood of local tissue inflammation.

Production of Reactive Oxygen Intermediates (ROIs)

Reactive oxygen intermediates (ROIs) are important metabolites in the capacity of macrophages and other inflammatory cells to kill invading bacteria and to attack cancer cells. However, increased overall production or inappropriate triggering of ROI release by macrophages can be a major contributor to tissue damage and the oxidation of cell surface lipids as well as DNA. The latter is one mechanism through which improperly regulated macrophages can actually increase the incidence of cancer. Results from many studies suggest that lead-exposure of macrophages can increase the release of superoxide anion and/or hydrogen peroxide at least shortly after exposure. Key studies are summarized in Table AX5-9.6.

In a recent study on environmentally exposed children in Mexico, Pineada-Zavaleta et al. (2004) reported that production of superoxide anion by directly activated (interferon-gamma + LPS) monocytes was directly correlated with blood Pb level. This was in contrast with the effect of arsenic, which had a negative association. In other studies involving low levels of exposures, Zelikoff et al. (1993) demonstrated that rabbits exposed to Pb via inhalation had pulmonary macrophages that produced elevated levels of both H_2O_2 and superoxide anion upon stimulation in vitro. In an in vitro study, Shabani and Rabani (2000) reported that Pb nitrate exposure produced a dose dependent increase in superoxide anion by rat alveolar macrophages. Baykov et al. (1996) fed BALB/c mice dietary Pb and found that peritoneal macrophages had an increased spontaneous release of H_2O_2.

Other studies have reported no effects of Pb on superoxide anion production when a long recovery period was included following in vivo exposure (Miller et al., 1998) as well as negative effects of Pb on oxidative metabolism by certain macrophages or macrophage cell lines (Castranova et al., 1980; Hilbertz et al., 1986; Chen et al., 1997). These somewhat different results suggest that the subpopulations of macrophages examined (e.g., alveolar vs. splenic vs. peritoneal) and the timeframe of assessment relative to exposure may be important factors in the effect of Pb on ROI production.

The biological importance of increased ROI production by lead-exposed macrophages should not be underestimated. Fernadez-Cabezudo et al. (2003) demonstrated that the potent antioxidant, vitamin E could protect TO strain mice against some lead-induced immunosuppressive alterations. Hence, macrophage-associated oxidative damage following exposure to Pb may be a mitigating factor in nonlymphoid organ lead-induced pathologies.

Arachidonic Acid Content and Prostaglandin Production

Archidonic acid (AA) is a major surface component of many cells, including macrophages, and is the precursor of cyclooxygenase and lipoxygenase metabolites. As a result, the specific AA content of membranes and the capacity of macrophages to produce immunomodulatory metabolites from AA are important to overall health of the individual. One of the findings since 1986 concerning lead-induced modulation of macrophage function is the impact of Pb on PGE_2 production. One study (Knowles and Donaldson, 1990) reported that diets supplemented with Pb at 500 ppm and fed to chicks produced an increase in the percentage of AA included in cell membranes. Such an increase would be expected to raise the risk of overall inflammation.

Several groups have reported that Pb exposure increases macrophage production of the immunosuppressive metabolite PGE_2. Lee and Battles (1994) reported that mouse macrophages exposed to Pb (10 μM) in vitro had elevated basal PGE_2 production, but under some stimulatory conditions, had decreased production of PGE_2. When Knowles and Donaldson (1997) fed Pb to turkey poults in the diet at a level of 100 ppm, macrophage production of prostaglandin F2 (PGF_2), PGE_2 and thromboxane production were all significantly elevated vs. the control. Flohe et al. (2002) showed that exposure of mouse bone marrow-derived macrophages to Pb-chloride resulted in increased production of PGE_2 that correlated with increased mRNA production for the necessary enzyme, prostaglandin H synthase type-2.

Tissue Homeostasis

In an important observation reflecting the impact of lead-induced immunotoxicity on nonlymphoid tissues, Pace et al. (2005) showed that neonatal exposure of mice to Pb-acetate via drinking water (0.1 ppm for 6 weeks, both through maternal nursing and direct) produced a significant reduction in the testicular macrophage population. This correlated with increased

estradiol levels in the testis and reduced male reproductive performance. The authors hypothesized that lead-induced alteration among testicular macrophages is linked to an impaired tissue environment that likely includes increased oxidative stress, apoptotic somatic cells, and reduced fertility of males.

Colony Formation and Population Distribution

The ability of bone marrow-derived macrophages (BMDM) to form colonies in response to certain growth factors (e.g., colony stimulating factor-1 [CSF-1]) is a property related to the growth and differentiation of subsequent macrophage populations. Kowelenko et al. (1991) found that exposure to CBA/J female mice to Pb-acetate (0.4 mM in drinking water for 2 weeks) reduced colony formation of macrophages in response to CSF-1. Infection of the mice with *Listeria* only exacerbated this effect of lead. The same authors (Kowelenko et al., 1989) had previously demonstrated that when BMDM were cultured in vitro with Pb-chloride (0.1 μM), colony formation was significantly impaired. These combined results suggest that exposure to Pb can impair the generation of macrophage populations as well as modulate the functional spectrum of fully matured macrophages. Bunn et al. (2001a) reported that gestational exposure of CD rats to 50 ppm Pb-acetate via the drinking water of the dams resulted in female adult offspring with a significantly decreased percentage (58% reduced) of circulating monocytes. A 100 ppm dose of Pb-acetate produced a significant reduction (74% reduced) in the absolute numbers of monocytes as well. The blood lead level at birth associated with the decreased percentage of macrophages in the adult offspring was 8.2 μg/dL. In general agreement, Lee et al. (2002) reported a significant decrease in the absolute numbers of circulating monocytes and polymorphonuclear leukocytes (PMNs) in juvenile female chickens exposed in ovo on embryonic day (E) 12 to 200 μg Pb-acetate. The corresponding blood lead level at hatching was 11.0 μg/dL. However, in this case, the lead-induced reduction in monocytes and PMNs was only seen in concert with an airway viral infection (viral stressor) and not in the resting uninfected animal.

Antigen Presentation and Lymphoid Stimulation

Exposure to Pb influences the interaction between macrophages and T lymphocytes, and as a result, the capacity of macrophages to support T lymphocyte proliferation and activation can

be altered as well. Kowelenko et al. (1988) found that mouse macrophages exposed to Pb (both in vivo and in vitro) can induce an increased proliferative response of T lymphocytes in co-culture but that antigen-specific stimulation of primed T cells is significantly reduced. Lead-suppressed antigen presentation capabilities of mouse macrophages were also reported by both Smith and Lawrence (1988) and Blakley and Archer (1981).

Chemotaxis

Chemotactic activity of macrophages is an important function required for the directed migration of macrophages to sites of infection and tumor growth. However, it is a functional capacity that has not been systematically examined within the lead-immune literature. Using female Moen-Chase guinea pigs, Kiremidjian-Schumacher et al. (1981) showed that Pb chloride exposure of peritoneal macrophages in vitro (10-6 μM) inhibited the electrophoretic mobility of the cells.

Phagocytosis and Clearance of Particles

Phagocytosis of targets and removal/clearance of dead cells and particles are major functions of macrophages. However, phagocytosis can involve a variety of different cell surface receptors on macrophages, depending upon both the nature of the target encountered and the subpopulation of macrophages examined. In general, phagocytic capacity of macrophages seems to be relatively insensitive to lead-induced immunomodulation compared with the effects on NO and TNF-α production.

However, differences in outcome in phagocytosis evaluations are likely to be based on the differences in the source of macrophages used and their relative activation state at the time of assessment. A few studies have described significant effects on phagocytosis, but these have usually relied upon phagocytosis mediated through the Fc receptor on macrophages. Because cell adherence to surfaces may be influenced negatively by Pb (Sengupta and Bishali, 2002), impairment of phagocytosis may also involve some lack in efficiency with macrophage anchoring to substrates. De Guise et al. (2000) reported no effect on bovine macrophage phagocytosis of latex beads by Pb at in vitro treatment concentration of 104 M. This was in contrast with suppressive effects of both cadmium and mercury. Using Sephadex-elicited peritoneal macrophages derived from young turkeys fed 100 ppm Pb in the diet, Knowles and

Donaldson (1997) found a 50% reduction in the percentage of phagocytic macrophages using SRBC targets. The activity per phagocytic macrophage was also reduced.

Kowolenko et al. (1988) studied the effect of Pb-acetate at 10 mM in the drinking water of CBA/J mice. They reported no effect on phagocytosis of *Listeria monocytogenes* targets, yet they found an overall decreased resistance to *Listeria.* When the same investigators exposed peritoneal and splenic macrophages to Pb in vitro (100 μM), they also found no significant effect of Pb on phagocytic activity. Jian et al. (1985) reported that New Zealand white rabbit-derived alveolar macrophages exposed to Pb in vitro at 10^{-5} M concentration were significantly impaired in the phagocytosis of opsonized chicken erythrocytes (Fc receptor-mediated phagocytosis). Trejo et al. (1972) reported that a single i.v. injection of Pb (5 mg/rat) into male Sprague Dawley (SD) strain rats produced an inhibition in the phagocytic capacity of Kupffer cells.

Several studies have reported a decreased clearance capacity of the reticuloendothelial system following in vivo exposure to lead. Filkins and Buchanan (1973) found that injection of 5 mg of Pb-acetate i.v. into male Holtzman strain rats produced reduced carbon clearance. Similarly, Trejo et al. (1972) reported that a single i.v. injection of Pb (2.5 mg) into male SD strain rats significantly reduced clearance of colloidal carbon.

In contrast, Schlick and Friedberg (1981) found that 20 μg/kg Pb-acetate in a single i.p. injection of NMRI strain mice significantly increased the clearance of India ink. Ironically, oral administration of Pb for 10, but not 30, days of 10 μg/kg resulted in an increase in clearance activity. Difference in route of Pb administration may be a factor in the different results obtained.

Induction of Heat Shock Proteins

One study (Miller and Qureshi, 1992), using a macrophage cell line, reported that exposure of macrophages (MQ-NCSU) in culture to Pb-acetate (1000 μM) induced the same set of four heat shock proteins as when the macrophages were subjected to thermal stress. This result fits the hypothesis that Pb produces a profound immunomodulatory effect in macrophages that has similarities with the exposure of macrophages to certain pathogens.

Apoptosis

Significant differences exist in the literature concerning the potential role of Pb in the apoptosis of macrophages. The difference may be based on the exposure methodologies (in vivo vs. in vitro) as well as the source of macrophages utilized. De la Fuente et al. (2002) found that human monocytes exposed to Pb in vitro at high concentrations did not undergo apoptosis. This was in direct contrast with the apoptosis-promoting effects of cadmium in the same assessment protocol. In contrast, Shabani and Rabibani (2000) exposed rat alveolar macrophage to Pb nitrate in vitro and found that 60 μM concentration produced a significant increase (2x) in DNA fragmentation after 3 to 24 h in culture.

5.9.7 Granulocytes and Natural Killer (NK) Cells

Other cell types important in innate immunity, as well as in immunoregulation, are the lymphoid population of natural killer cells and granulocytes, including PMNs (i.e., neutrophils). Neither population appears to be a major target for lead-induced immunotoxicity, although both may be influenced indirectly via immune cell-cell interactions as well as by changes in cytokine production. Among the two, neutrophils may be the more sensitive cell type based on assays conducted to date. For neutrophils, several groups have reported alteration in chemotactic activity following exposure to lead. Queiroz et al. (1993) found impaired migration ability of neutrophils from battery workers occupationally exposed to lead. Likewise, Valentino et al. (1991) had a similar observation among male occupationally exposed workers. Lead exposure of young SD strain rats can increase the population of neutrophils (Villagra et al., 1997), although, as the authors indicated, this does not necessarily afford enhanced host protection against disease. Baginski and Grube (1991) reported that human neutrophils exposed to Pb had increased killing capacity, probably via increased release of ROIs despite having reduced phagocytic capacity. This would fit the same general profile as the effects of Pb on macrophages. Therefore, neutrophils may contribute to lead-induced tissue inflammation and damage via increased ROI release. Yet, their effectiveness in protection against disease challenge may be no greater following exposure to Pb, because some impairment in chemotaxis and phagocytosis has been reported as well.

Yucesoy et al. (1997) reported that either Pb exposure or simultaneous exposure to Pb and cadmium in human workers did not impair NK cytotoxicity activity. This finding was supported

by studies using in vivo exposure to Pb in rats (Kimber et al., 1986) and mice (Neilan et al., 1983). Therefore, it would appear that NK cells are not a prime target associated with lead-induced immunotoxicity, although more subtle effects may certainly exist within the cell type.

Eosinophils represent an important granulocytic cell type in type 2 associated inflammatory and allergic reactions. However, few studies have examined Pb exposure and eosinophil activity. Villagra et al. (1997) reported that exposure of female juvenile SD rats to Pb [four alternate-day s.c. injections of 172 mg/g body wt Pb-acetate] increased the degranulation of eosinophils (in animals given estrogen 1 day later). Such a response would be expected to contribute to increased inflammation.

5.9.8 Hypersensitivity and Autoimmunity

At the time of preparation of the 1986 AQCD, little was known about the potential for Pb to influence the risk of allergic and autoimmune diseases. However, since the early 1990s, a significant number of studies have all pointed toward the fact that Pb causes a profound dysregulation of the immune system. It skews the balance of responses in directions that reduce certain host defenses against infectious diseases while enhancing the risk of allergic and autoimmune disease. Lead exposure at low to moderate levels appears to alter T lymphocyte responses in such a way as to increase the risk of atopy, asthma, and some forms of autoimmunity. Increased IgE production following exposure to Pb is among the most frequently reported immune alterations. Elevated IgE levels would be an associated risk factor for atopy and allergic disease. Several investigators have discussed the fact that Pb is a likely risk factor associated with the increased incidence of childhood allergic asthma (Miller et al., 1998; Heo et al., 1998; Snyder et al., 2000; McCabe et al., 2001; Dietert et al., 2004; Transande and Thurston, 2005) as well as later life allergic disease (Heo et al., 2004). Joseph et al. (2005) observed no association for childhood BLL and risk of asthma among an African-American population. However, results on other populations from this study, including those involving Caucasian children with BLLs above 5 µg/dL, led the authors to call for further studies into the possible linkage of early life lead exposure and risk of asthma (Joseph et al., 2005).

As described by McCabe et al. (1991) and discussed by Dietert et al. (2004), lead-induced immunotoxicity is novel in that profound cellular toxicity is not evident following exposure at low to moderate exposure concentrations. In fact, antibody responses overall are usually

unaffected or may be increased depending upon the class/isotype measured. However, the functional responses mounted following Pb exposure do not reflect the normal immune balance that would otherwise occur. This dysregulation can alter the risk of certain autoimmune diseases based on several observations. Holladay (1999) has considered the importance of the timing of exposure and the fact that early life exposure may establish the immune profile that then contributes to later disease including autoimmunity.

Hudson et al. (2003) reported that exposure to Pb can exacerbate systemic lupus erythmatosus (SLE) in lupus-prone strains of mice. In contrast with the effect of mercury, these authors found that for lupus, Pb exposure would not induce this autoimmune condition in genetically resistant mice but would increase severity of the disease in genetically prone animals. The authors noted some gender effects within certain strains (e.g., NZM88). Using early in ovo exposure to Pb (10 μg/egg), Bunn et al. (2000) found that Pb-acetate-exposed male chicks could be induced to produce autoantibodies against thyroglobulin, which were not present in acetate-exposed controls. No lead-induced alteration was observed in females that were predisposed to mount anti-thyroglobulin responses. The gender effect is intriguing in that autoimmune thyroiditis in genetically predisposed strains is always more severe in females than in males.

Two lines of evidence suggest that the capacity of Pb to influence the risk of autoimmunity is not always associated with simply a strict shift from Th1 to Th2 responses. Hudson et al. (2003) discussed the fact that lupus is not purely a Th2-mediated disease, but rather seems to occur under conditions associated with skewing in either direction. McCabe et al. (2001) found that Pb can increase the stimulation of alloantigen reactive T cells (where macrophage processing of antigen is required) but not enhancement of T cell clonotypic responses against either mitogens or superantigens (where processing is not required). This suggests that the role of Pb in influencing risk of autoimmune disease goes beyond a simple consideration of Th1/Th2 balance. In fact, Goebel et al. (2000), studying mucosal immunity, reported that administration of Pb-chloride to NOD strain mice produced a gut cytokine microenvironment that was skewed toward Th2 over the short run, but later was shifted toward Th1 with increased production of IFN-γ. This shift to Th1 was accompanied by a loss of tolerance and capacity to mount an immune response against a diet-associated protein (chicken ovalbumin). The authors proposed that reduction of the capacity for oral tolerance would predispose an individual toward autoimmune disease.

Finally, Waterman et al. (1994) and El-Fawal et al. (1999) have described the production of autoantibodies against neural proteins in both battery workers and rats exposed to low levels of Pb via drinking water. These authors have suggested that exposure to Pb may precipitate the autoimmunity by altering antigen immunogenicity and/or the capacity of the immune system to respond to certain antigens. This, in turn, may contribute to the eventual lead-associated neurological disease.

5.9.9 Mechanism of Lead-Based Immunomodulation

In the 1986 AQCD, there was little direct information available about the immune system regarding the molecular mechanism(s) of lead-induced immunotoxicity. Binding to thiol groups and altering cell surface receptors were indicated as possible factors in altered immune function. Since that time, some additional information has been generated through a variety of studies on human and animal immune cells. However, a clear or simple explanation remains to be determined. Table 5-9.3 lists studies on the immune system that have contributed to a better understanding of potential mechanisms or have forwarded potential hypotheses with some supporting data.

At the level of cell-cell interactions, it seems clear that Pb alters metabolism and cytokine production by macrophages and antigen presenting cells. It also reduces their capacity to respond to growth factors such as CSF-1 (Kowelenko et al., 1989). Pace et al. (2005) discussed the hypothesis that reduced populations of functionally altered macrophages (because of lead-induced unresponsiveness to CSF-1and over production of ROIs) in tissues can produce nonimmune problems. The model they used is the homeostatic presence of testicular macrophages and the likelihood that lead-induced macrophage immunotoxicity contributes directly to lead-associated reduction in male fertility.

Additionally, Pb is known to selectively alter cell signaling to CD4+ T cell subpopulations, promoting proliferation in some but not others. The outcome is enhanced tissue inflammation, reduced CMI, and increased production of atopy-inducing antibodies. Risk of autoimmune reactions is increased in some models of lead-induced immunotoxicity. For example, Heo et al. (1997) reported that lead-exposed murine T lymphocytes are biased in expression of Vβ genes. This is potentially problematic as this phenotype is common among a

Table 5-9.3. Suggested Mechanisms of Lead-Induced Immunotoxicity

Species	Strain/Gender	Suggested Endpoints	Associated Functional Alteration	Lowest Effective Dose	Duration	References
Mouse	Balb/c	CSF-1 Responsiveness of Macrophages	↓Testicular macrophages ↓Fertility	0.1 ppm	6 weeks	Pace et al. (2005)
Mouse	TO strain males	Vitamin E protection against lead-induced splenomegaly	↑Putative ROI associated splenomegaly	1 mg/kg	2 weeks	Fernandez-Cabezudo et al. (2003)
Chicken	Cornell K Strain	Thymulin partial reversal of Th skewing	↓Lead-induced DTH suppression	400 µg	Single in ovo injection	Lee and Dietert, (2003)
Mouse	Balb/c females C57 Bl/6 females	Lead disruption of antigen processing and presentation signals	↑Alloreactive $CD4+^{high}$ cells ↑Risk of autoimmunity	0.5 µM in vitro	4 days	McCabe et al. (2001)
Mouse	C 57Bl/6	PKC activation	↑TNF-α, ↑IL-6 ↑PGE_2	20µM in vitro	4.5 hrs	Flohe et al. (2002)
Rat	PC-12 cells	NF-κB activation AP-1 induction C-Jun kinase induction	↑ROI	1 µM in vitro	5-120 min	Ramesh et al. (1999)
Mouse	DO11.10 ova-mice	Adenylcyclase activation with elevated cAMP levels	↑Th skewing	2.5 µM in vitro	15 mins-6 hrs	Heo et al. (1998)
Mouse	DO11.10 ova-tg mice	Vβ gene usage	↑Risk of autoimmunity	50 µg 2x/week s.c.	8 weeks	Heo et al. (1997)
Human	-	NF-κB activation in CD4+ cells	↑Risk of autoimmunity and hypersensitivity	1 µM	30 min	Pyatt et al. (1996)
Mouse	CBA/J females	↑Immunogenicity of neural proteins	↑Autoimmune mediated neurological damage	Lead-altered proteins used as antigens	3 injections of lead-modified neural proteins	Waterman, et al. (1994)
Mouse	Swiss Females	↑TNF-α production	↑Sensitivity to endotoxin	5 mg	Single i.p. injection	Dentener et al. (1989)

variety of human and animal model autoimmune conditions. A variety of exogenous factors have been reported to partially ameliorate the immunotoxic effects of lead. Chelation of Pb in lead-exposed dams corrected some lead-induced immunotoxic problems in the rat female offspring, but it left the animals with some DMSA-induced immune alterations (Chen et al., 1999). Other exogenously administered factors that have been reported to partially restore lead-suppressed immune function are vitamin E (Fernandez-Carbezudo et al., 2003) and thymulin (Lee and Dietert, 2003).

At the subcellular level, the bases for immunotoxic changes remain speculative. McCabe et al. (2001) suggested that altered antigen processing and subsequent cell signaling to T cells may be an explanation for the capacity of Pb to selectively increase CD4+ (high density) cells. Certainly Pb appears to alter signal transduction. It appears to elevate expression of the nuclear transcription factor NF-κB (Pyatt et al., 1996; Ramesh et al., 1999) as well as increase expression of AP-1 and cJun (Ramesh et al., 1999). Flohe et al. (2002) found evidence that Pb can elevate the activation of PKC. The authors speculated that this might be involved in lead-induced increases in TNF-α production. Additionally, Heo et al. (1998) reported that Pb increases adenyl cyclase activity among T lymphocytes, generating elevated cAMP levels. The authors hypothesized that this effect, in conjunction with differences in cell signaling pathways for promoting Th1 vs. Th2 cells, may be involved in the capacity of Pb to skew Th0 helper cells toward Th2.

5.9.10 Age-Based Differences in Sensitivity

With the literature available at the time of the 1986 AQCD, it was virtually impossible to evaluate age-based differences in susceptibility to lead-induced immunotoxicity. However, in recent years, this has become a major topic of study for many toxicants including lead. Several studies have added to the available data assessing the developmental immunotoxicity of Pb (reviewed in Barnett [1996], Dietert et al. [2000, 2004], Lee and Dietert [2005]). Several patterns have emerged from exposure data using animals of different ages.

First, it seems clear that blood Pb levels at or near birth of below 10 µg/dL can be associated with juvenile and/or adult immunotoxicity. Several studies reported effects in the range of 5-8 µg/dL. These low levels would seem to place the immune system on par with the

neurological system in terms of potential sensitivity to lead. Table 5-9.4 shows examples of studies in which low blood lead levels were linked with immunotoxicity.

Table 5-9.4. Immunomodulation Associated with Low Blood Lead Levels in Animals

Species	Blood lead (µg/dL)	Age at Measurement	Immune Parameter(s)	Age at Assessment	Reference
Mouse	~5.0	1 week	↑IgE, ↓ Splenic T Cell Populations	2 weeks	Snyder et al. (2000)
Rat	8.2	1 day	↓monocytes	13 weeks	Bunn et al. (2001a)
Rat	6.75	4 weeks	↓DTH, ↓IFN-γ, ↑IL-4	13 weeks	Chen et al. (2004)
Rat	8.0	4 weeks	↑TNF-α ↑Rel. Spleen weight	13 weeks	Lee et al. (2002)
Chicken	8.2	1 day	↓circulating lymphocytes post infection	5 weeks	Lee et al. (2002)
Chicken	11.0	1 day	↓DTH and ↓TLC, monocytes, PMNs post infection	5 weeks	Lee et al. (2001)
Chicken	7.0	1 day	↑autoantibody production	10 weeks	Bunn et al. (2000)

A second finding is that the immunotoxic effects induced by Pb are persistent long after blood levels and potential body burdens of Pb are significantly reduced. Miller et al. (1998), Chen et al. (1999), Snyder et al. (2000), and Lee et al. (2001) all emphasize this latter point. In fact, in most of these studies immunotoxic alterations were present when Pb levels in exposed animals were not distinguishable from control levels. This should provide a cautionary note regarding studies in humans. Data from adult exposures provides little insight into the potential persistence following adult exposure to lead. However, rather than the developing immune system being more regenerative postexposure and able to withstand immunotoxic insult, it appears that the non-dispersed developing immune system is a particularly susceptible target to many immunotoxicants (Dietert et al., 2002).

A third, and somewhat surprising, finding concerning early exposure to Pb is that qualitative differences in the spectrum of immune alterations can exist, depending upon the

developmental window of exposure. Figure 5-9.1 illustrates this point. Early embryonic exposure of rats and chickens to Pb failed to alter juvenile DTH responses, despite significant effects on macrophage function. However, exposure to Pb after the mid-embryonic point of embryonic development readily suppressed subsequent DTH. As shown in Figure 5-9.1, the development window in which sensitivity to DTH suppression emerges is quite similar in the two species. This observation suggests that both quantitative (LOAELs) and qualitative (range of immune alterations) differences in sensitivity to Pb can exist across different age groups.

Additionally, some studies in animals have noted gender differences in the effects of Pb following exposure (Bunn et al., 2000, 2001a,b, c; Hudson et al., 2003). Gender differences have also extended to results in humans as per lead-induced immune and inflammatory alterations (Karmaus et al., 2005; Fortoul et al., 2005). It seems feasible that, even in the embryo, hormonal differences among females and males may impact some outcomes of low-level Pb exposure.

Table 5-9.5 shows comparisons of the lowest reported blood Pb levels at different ages associated with the same immunotoxic endpoint. From these limited comparisons, it would appear that different ages of rodents (e.g., embryonic vs. adult) differ in dose sensitivity for lead-induced immunotoxicity somewhere in the range of 3 to 12-fold. Clearly, additional direct comparisons would help to refine this estimate.

A fourth observation from the early exposure studies is that exposure to even very low levels of Pb can predispose the immune system for unanticipated postnatal responses when the system is stressed. This general phenomenon is called latency. Lee et al. (2002) provided an example of this following the single in ovo exposure of embryonic day 5 chick embryos to low levels of Pb (10 μg; blood lead level 1 day post hatch of 8.2 μg/dL). The leukocyte profiles of the animals appeared to be completely normal. However, when these animals were exposed to a respiratory virus, their pattern of leukocyte mobilization was completely aberrant from controls. Therefore, some immunotoxic alterations following early exposure to low levels of Pb may only be evident during periods of postnatal stress.

Several studies have reported the positive association of blood Pb levels in children with elevated serum IgE (Karmaus et al., 2005; Sun et al., 2003; Lutz et al., 1999). These observations are supported by the animal data in rats and mice (Miller et al., 1998; Snyder et al., 2000) and suggest that lead-induced risk of atopy and asthma may be a particular health issue.

Table 5-9.5. Comparisons of Age-Based Sensitivity to Lead-Induced Immunotoxicity

Species	Altered Endpoint	Embryo – fetal*	Neonatal*	Adult*	References
Mouse	↑IgE	~5µg/dL	12 µg/dL	38 µg/dL	Snyder et al. (2000) Heo et al. (1996)
Rat	↓DTH (persistent effect assessed 13 weeks post-exposure)	34 µg/dL	-	>112 µg/dL (measured at birth for persistent effect)	Miller et al. (1998) Bunn et al. (2001b)
Mouse	↓DTH	-	29 µg/dL	87 µg/dL	Faith et al. (1979) McCabe et al. (1999)
Rat	↑TNF - α (persistent effect assessed 13 weeks post-exposure)	8 µg/dL	–	>112 µg/dL (measured at birth for persistent effect)	Miller et al. (1998) Chen et al. (2004)

* Lowest blood lead concentration reported with effect

Trasande et al. (2005) recently discussed the fact that, despite progress in reducing the deposition of Pb in the environment, Pb continues to be a concern relative to asthma and children's health.

5.9.11 Summary and Conclusions

The immune system appears to be one of the more sensitive systems to the toxic effects of lead. The 1986 AQCD provided an excellent summary of the studies that had been conducted prior to that date. But knowledge of fundamental immunology has progressed greatly during the past 20 years. Not surprisingly, the large number of studies conducted since the mid-1980s provided a much clearer understanding of the immune-associated problems that can arise from problematic exposure to lead. Studies across humans and a variety of animal models are in general agreement concerning both the nature of the immunotoxicity induced by Pb as well as the exposure conditions that are required to produce immunomodulation. Figure 5-9.2 summarizes the basic immunotoxic changes induced by Pb that result in Th skewing, impaired macrophage function, and increased risk of inflammation-associated tissue damage.

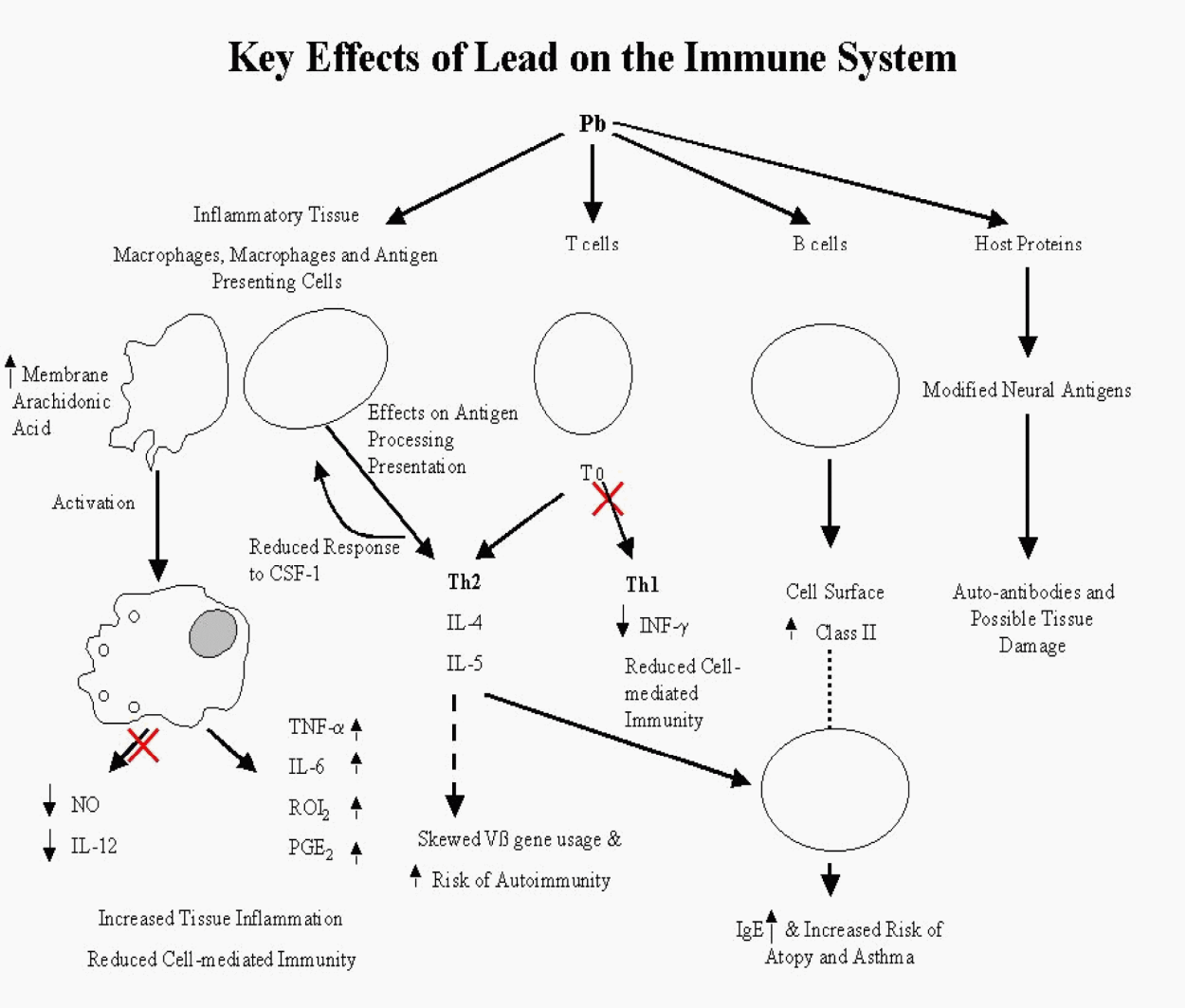

Figure 5-9.2. This figure shows the fundamental alterations to the immune system and to immunological response and recognition induced by exposure to lead. The functional shifts are disproportionate compared to the relatively modest changes among leukocytes with low to moderate exposure to lead.

Lead is unlike many immunotoxicants in that, at low to moderate levels of exposure, it does not produce overt cellular cytotoxicity or lymphoid organ pathology. However, it can induce profound functional alterations that influence risk of disease. Lead preferentially targets macrophages and T lymphocytes, although effects have been reported in B cells and neutrophils as well. There are three major hallmarks of lead-induced immunotoxicity. First, Pb can dramatically suppress the Th1-dependent DTH response, as well as production of associated Th1

cytokines. Second, Pb can dramatically elevate production of IgE while increasing production of Th2 cytokines, such as IL-4. Third, and perhaps most sensitive, is the modulation of macrophages by Pb into a hyperinflammatory phenotype. After exposure to lead, macrophages significantly increase production of the proinflammatory cytokines TNF-α and IL-6 (and in some studies IL-1). Many studies also reported elevated release of ROIs and prostaglandins. Ironically, production of one of the most important host defense factors, NO, is consistently and severely suppressed by exposure to lead. This package of lead-induced changes among macrophages makes them more prone to promote tissue destruction but actually less capable of killing bacteria or possibly presenting antigens to T lymphocytes. The Pb-induced shift in phenotype explains the capacity of inhaled Pb to promote bronchial inflammation while bacterial resistance is severely depressed.

Lead-induced skewing of Th activity (biasing responses toward Th2) across a population would lead to the expectation of a greater risk of atopy, asthma, and some forms of autoimmunity. Concomitantly, resistance to some infectious diseases could be reduced. This predicted change of risk might help explain some recent trends in the incidence of diseases, such as the epidemic rise in allergy and some forms of asthma in the United States.

Sensitivity of the immune system to Pb appears to differ across life stages. Studies in rats and mice suggest that the gestation period is the most sensitive life stage followed by the early neonatal stage. But even during embryonic, fetal, and early neonatal development, critical windows of vulnerability are likely to exist. Compared to adults, the increased dose sensitivity of the embryo-fetus would appear to fall in the range of 3-10x depending upon the immune endpoint considered. Some studies have found evidence for gender differences in the impact of Pb on the immune system particularly with early life exposures. Potential gender differences in immunotoxic outcome may be important in the evaluation of those populations at greatest risk.

Recent studies have suggested that exposure of embryos to Pb producing neonatal blood lead concentrations below 10 μg/dL can also produce later-life immunotoxicity (see Table 5-9.4). Furthermore, immunotoxicity persists long after any evidence of prior embryonic Pb exposure. This latter observation from several laboratories may have implications for the design of human epidemiological studies.

5.10 EFFECTS OF LEAD ON OTHER ORGAN SYSTEMS

In the 1986 Pb AQCD, the discussion of other organ systems included cardiovascular, hepatic, gastrointestinal (GI), and endocrine systems. Due to our increased understanding on the effects of Pb on cardiovascular and renal systems and their contribution to potential health effects of Pb, separate sections (5.5, 5.7) were dedicated earlier in this chapter to detailed discussions on these aspects. Similarly, with our increased understanding on the effects of Pb on endocrine functions and its inherent role with respect to neurotoxicological, reproductive, and developmental effects, literature reviewed for Pb effects on the endocrine system is discussed in the respective sections. This section focuses on the discussion of Pb effects on the hepatic and GI systems.

5.10.1 Effects of Lead on the Hepatic System

The liver is a highly active metabolic tissue. Apart from its roles in fatty acid metabolism and limited heme synthesis function, the liver also has a major role in guarding other systems from the toxic effects of xenobiotic compounds using a huge complement of detoxification machinery referred to as phase I and phase II enzyme systems. Limited studies on experimental animals reported in the 1986 Pb AQCD indicated that Pb induced effects in the hepatic system. Laboratory animals, especially rats, exposed to Pb-nitrate have exhibited increased liver cell proliferation, DNA synthesis, cholesterol synthesis, and glucose -6-phosphate dehydrogenase (G6PD) activity indicative of Pb-induced hyperplasia. Further, the literature reviewed in the 1986 Pb AQCD reported alterations in the levels of drug metabolizing enzymes in experimental animals given large doses of Pb. The evidence for such effects in humans was less consistent. The 1986 document also concluded that the effects on the liver occurred only at high exposure levels. The majority of studies on the effects of Pb on the hepatic system in experimental animals that are reviewed in this document report functional and biochemical changes in the liver, clearly pointing to metabolic perturbations in liver. For ease in understanding and integration of these functional changes, the discussion is divided into the following four subsections: hepatic drug metabolism, lipid and glycogen metabolism and lipid peroxidation, heme synthesis, and toxicity mitigation by chelation and other interventions.

5.10.1.1 Hepatic Drug Metabolism

Approximately 75% of the hepatic blood comes directly from the gastrointestinal viscera, with the majority of drugs or xenobiotics absorbed coming directly to the liver in concentrated form. The liver is equipped with a huge complement of drug metabolizing enzymes that detoxify many of the xenobiotics but also activate the toxicity of others. Oxidation and conjugation of xenobiotics have historically been referred to as phase I and phase II reactions. The phase I enzymes include cytochrome P450 (CYP450) heme-containing monoxygenases, flavin-containing monoxygenases, and epoxide hydrolases. The phase II enzymes include glutathione (GSH) S-transferases (GST), UDP-glucuronyl transferases (UGT), N-acetyltransferases (NAT), and sulfotransferases (SULT). Xenobiotic metabolism by these two complements of enzyme systems are essential for catabolizing and eliminating of drugs; however, this process can also produce activated toxicants and carcinogens. A limited number of these CYP450s are involved in the biosynthetic pathways of steroid and bile acid production. It has been increasingly recognized that, under certain circumstances, CYP P450s can produce ROS that result in oxidative stress and cell death.

Liver is an active tissue. In addition to xenobiotic metabolism, it also participates in gluconeogenesis, fatty acid metabolism, and cholesterol biosynthesis. Research concerning the effects of Pb on the hepatic system in the past 15 years has provided some preliminary indications of Pb-induced alterations in many of the hepatic functions described above. The following discussion presents, as much as possible, the effects of Pb on individual enzymes, but due to the multifarious interactions of many of these metabolic enzymes, there may be places such separation was not possible.

Phase I Enzyme

Earlier studies on the toxic effects of Pb on hepatic drug metabolizing enzymes demonstrated that acute exposure to Pb-acetate decreased rat hepatic CYP450s with increased levels of urinary δ-aminolevulinic acid (ALA). Co-treatment with phenobarbitol, a CYP450 inducer, was shown to reverse the decrease CYP450 levels, suggesting a Pb-acetate-mediated inhibition of heme synthetic enzymes. Decreased activities of estradiol-17 beta enzyme observed in rat liver treated with triethyl Pb-chloride (Odenbro and Arhenius, 1984) suggest that both Pb and organo-Pb compounds are capable of inhibiting CYP450 activities. Roomi et al.

(1986) also observed decreased levels of hepatic microsomal CYP450s and decreased aminopyrene-N-demethylase activity on exposure to a single dose of Pb-nitrate (5-10 mmol/kg body wt). This decrease in phase I enzymes was followed by increased levels of phase II components such as GSH, GST, and DT diaphorase, suggesting that Pb-nitrate and Pb compounds can induce biochemical properties characteristic of hepatocyte nodules. Subchronic (2-3 months) exposure to Pb-acetate (5-50 mg/kg body wt) had been found to induce CYP450s and cytochrome b5 in rat liver and kidney (Nehru and Kaushal, 1992). As described earlier, multiple isoforms of CYP450s exist in the liver.

To identify the inhibitory effect of acute Pb exposure on specific isoform(s), Degawa et al. (1994) exposed male F344 rats to Pb nitrate (20,100 μmol/kg body wt) and evaluated liver CYP450s 24 h postexposure. Lead-nitrate exposure preferentially inhibited cytochrome P4501A2 enzyme activity in liver microsomal preparations as assayed for mutagenic conversion of substrates 2-amino-6-methyl-dipyridol [1,2-a; 3',2-d] imidazole and 3-amino-1-methyl-5H-pyridol[4,3,-b]indole. Lead-nitrate exposure also inhibited the induction of cytochrome P4501A2 by the inducers 3-methylcholanthrene and 2-methoxy-4-aminoazobenzene at both the protein and mRNA levels. The authors further concluded that the specific inhibition of P4501A2 by Pb-nitrate observed may have been due to inhibition of heme synthesis, as Pb-nitrate was not found to inhibit P4501A2 activity in vitro. Additional studies carried out by the same group using various metal ions (e.g., Pb, Ni, Co, and Cd) found that the specific inhibition of P4501A2 was unique to Pb-nitrate (Degawa et al., 1994, 1995). Degawa et al. (1996) also investigated the effect of Pb-nitrate-mediated inhibition of CYP1A gene activity in rat liver by specific inducers and reported that Pb-nitrate inhibited the induction of CYP1A mRNA by aromatic amines, but not by aryl hydrocarbons, suggesting the role of other cellular factors in the transcriptional activation of CYP1A genes. Lead-nitrate has been reported to induce the production of TNF-α in rat liver (Shinozuka et al., 1994), a cytokine implicated in the suppression of constitutive expression of CYP1A2 mRNA in rat hepatocytes. Based on these findings, Degawa et al. (1996) concluded that the inhibition of constitutive and aromatic amine-induced expression of CYP1A2 in rat liver caused by Pb-nitrate may occur at least in part by TNF-α-associated mechanisms. Lead-nitrate (0.33 mg/kg body wt) pretreatment-mediated protection conferred against carbon tetrachloride (0.3 mL/kg)-induced hepatotoxicity as reported by Calabrese et al. (1995) may be due to the inhibition of CYP450 activities in liver by Pb.

Jover et al. (1996) investigated the effect of heme deficiency on Pb-induced hepatic P450 function and transcription. These authors concluded that the decrease in hepatic P450 resulting from Pb intoxication was mediated by two different mechanisms. One mechanism is involved inhibitory effects on P450 by Pb at the transcriptional level; the second was heme- dependent, as Pb-mediated inhibition of heme synthesis decreased the heme saturation of P450 and the apo-P450 ratio.

The effect of heavy metals (Cd, Co, Cu, Ni, Pb, and Zn) on 3-methylcholanthrene-induction of cytochrome P4501A and the activity of ethoxyresorufin-O-deethylase (EROD) were investigated in fish hepatoma cells (PLHC-1) by Brucshweiler et al. (1996). The authors reported that all the heavy metals tested had more pronounced effects on EROD activity compared to controls. The inhibitory potency of Pb was reported to be very low compared to cadmium or cobalt. A single treatment of Pb-acetate induced hepatic DT diaphorase activity (Sugiura et al., 1993). This induction of hepatic DT diaphorase by Pb-acetate has been reported to be decreased with concomitant administration of Dil, a calcium antagonist. Based on these observations, Arizono et al. (1996) suggested that DT diaphorase induction by Pb-acetate may occur de novo via protein synthesis mediated by increased cellular calcium. The potential interaction of metals, including Pb, on the induction of CYP1A1 and CYP1A2 by polycyclic aromatic hydrocarbons (PAHs) in human hepatocyte cultures was investigated by Vakharia et al. (2001). Lead-nitrate, like other metals such as Cd, Hg, and As, decreased the extent of CYP1A1 and CYP1A2 induction by five different PAHs. The authors concluded from these studies that Pb (5 μM) diminished the induction of CYP1A1 and CYP1A2 in human hepatocytes by ultimately decreasing the levels of CYP1A1 protein that was normally attainable through PAH induction. Korashy and El-Kadi (2004) also investigated similar interactions of metals with aryl hydrocarbon receptor (AHR)-regulated gene expression and enzyme activities in wild-type murine hepatoma cells (Hepa 1c1c7) and AHR-deficient cells (C12). These studies indicated that metals alone (including Pb) did not significantly alter CYP1A1 proteins or activity, or change AHR ligand-induced enzyme activity. There was no change in mRNA levels. Lead, in the presence or absence of AHR ligand, increased the activity of NAD(P)H:quinone oxidoreductase and its mRNA levels.

Phase II Enzymes

A single injection of Pb-nitrate (5-10 μM/100 g body wt) was found to increase GST activity levels (Roomi et al., 1986). Additional studies by the same group identified induction of a specific form GST-P by Pb-nitrate in rat liver (Roomi et al., 1987). Because a single injection of Pb-nitrate decreased phase I and increased phase II hepatic enzymes, these investigators concluded that Pb-nitrate treatment initiated a biochemical phenotype similar to carcinogen-induced hepatocyte nodules. Immunohistochemical analysis by the same group reported that Pb-nitrate administration resulted in the appearance of GST-P in most of the hepatocytes, an enzyme that is otherwise undetectable in normal rat liver (Columbano et al., 1988; Roomi et al., 1987). On the other hand, Nakagawa (1991) reported inhibition of GST on acute exposure to Pb and that the inhibition of GST followed a reduction in liver GSH levels. Nakagawa (1991) concluded that the depletion of GSH was not necessarily a critical factor in inhibiting GST.

Planas-Bohne and Elizdale (1992) reported that acute exposure to Pb-nitrate (100 μmol/kg) caused a significant increase in liver and kidney GST activity. Gel electrophoresis analysis to evaluate the contribution of various GST isoforms indicated that enhancement of liver GST activity was predominantly due to induction of GST isoform 7-7 in liver compared to all isoforms in kidney. Liver GST-P isoform was reported to be induced by both Pb-acetate and Pb-nitrate (Boyce and Mantle, 1993; Koo et al., 1994). This transient induction of GST-P has been regulated at transcription, post-transcription, and post-translational levels. Suzuki et al. (1996) utilized a transgenic approach to investigate the transcriptional regulation of GST-P induced by Pb and identified glutathione S-transferase P enhancer I (GPEI), an enhancer (whose core consists of two AP-1 site-like sequences) located at the 5' flanking region of this gene. The authors demonstrated that GPEI is an essential element in the activation of the GST-P by Pb and that the trans activating factor AP-1 is likely to be involved, at least in part, in the transcriptional activation of the GST-P gene by Pb via the GPEI sequence.

Daggett et al. (1997, 1998) investigated the effect of inorganic and organic Pb on liver GST expression and other phase II detoxifying enzymes in rat liver and kidney. Triethyl Pb chloride (TEL) injection (10 mg/kg body wt) decreased liver GST activity, as well as levels of various other GST isoforms (Daggett et al., 1997), in contrast to significant induction of kidney GST activity, suggesting that a single compound, TEL, had opposite effects on the expression of GST isozymes and indicated the complexity of GST regulation. Similarly, this group also

reported that a single injection of Pb-acetate (114 mg/kg body wt) reduced GSH levels, increased production of malondialdehyde (MDA), and did not change the expression of various GST isoforms analyzed, except GST-p1 on repeated injection (Daggett et al., 1998). Similar to studies with TEL, Pb-acetate also increased the expression of GST enzyme activity and expression of various isoforms without changing GSH and MDA levels, suggesting that oxidative stress may not be mediating the toxicity in kidney. On the other hand, TEL exposure was found to decrease microsomal estradiol metabolism (Odenbro and Rafter, 1988). The suppression of GST expression reported by Daggett et al. (1997, 1998) is in contrast to the induction of GST reported by various other groups discussed earlier. Other GSH-dependent enzymes (i.e., GSH peroxidase, GSH reductase) have been found to be suppressed with a simultaneous increase in oxidized GSH (GSSG) and a reduction in GSH/GSSG ratio (Sandhir and Gill, 1995). More detailed information on these and related studies is summarized in Table AX5-10.1.

5.10.1.2 Biochemical and Molecular Perturbations in Lead-Induced Liver Tissue Injury

Oskarsson and Hellström-Lindahl et al. (1989) studied the cellular transport of Pb (^{203}Pb), in rat hepatocytes using dithiocarbamate (DTC). Cells treated with Pb-acetate and Pb-DTC lipophylic complex demonstrated increased cytosolic Pb levels compared to Pb alone. This was further evaluated by measuring levels of ALAD. Cells treated with Pb-DTC complex showed rapid and stronger inhibition of ALAD compared to Pb-acetate, suggesting that this inhibition was due to increased mobilization of Pb into cells treated with Pb-DTC complex. Another report by the same group, Hellström-Lindahl and Oskarsson (1990), suggested that the increased inhibition of ALAD was due to the release of Pb from the Pb-DTC complex by decomposition. Using the mouse strain with a duplication of the ALAD gene (DBA), Claudio et al. (1997) reported increased accumulation of Pb in this strain by many fold as compared to mice with a single copy of the ALAD gene (C57).

A single injection of Pb-nitrate was reported to cause hepatic hyperplasia correlating with hepatic de novo synthesis of cholesterol along with alterations in glucose and lipid metabolism leading to altered serum lipid profiles (Dessi et al., 1984; Pani et al., 1984). Mobilization of hepatic glycogen and altered gluconeogenic enzymes, including differential expression of G6PD, have been reported following Pb exposure (Batetta et al., 1990; Hacker et al., 1990). Chronic Pb

intoxication has also been reported to inhibit gluconeogenic enzymes, alterations that were implicated in Pb bio-transformation rather than liver cell proliferation in Wistar rats (Calabrese and Baldwin, 1992). Increased levels of serum lipid peroxide (LPO) were also observed in rats given SC injection of Pb-acetate, supporting similar increased levels of serum LPO in humans exposed to Pb (Ito et al., 1985). These initial studies suggest that alterations in liver intermediary metabolism occur on exposure to Pb with a role for Pb-induced LPO in hepatotoxicity and potential involvement of oxidative stress in Pb toxicity.

Dessi et al. (1990) investigated the role of fasting on Pb-induced hepatic hyperplasia by monitoring the activities of enzymes involved in cholesterol synthesis and the hexose monophosphate shunt and reported that stimulation of these enzymes, even in Pb-acetate-treated fasting rats, supported the role of new endogenous synthesis of cholesterol and gluconeogenic mechanisms in Pb-induced hepatic cell proliferation. Chronic exposure to Pb was found to increase the arachidonate/linoleic acid ratio in liver and serum (Donaldson and Leeming, 1984; Donaldson et al., 1985) along with the GSG concentration (McGowan and Donaldson, 1987). As GSH and arachidonate are precursors for peptido-leukotrienes, Donaldson's group investigated the potential effects of dietary Pb on levels of fatty acids, peptido-leukotrienes, and arachidonate/linoleic ratios in chicken fed with diets low in calcium and methionine. These investigations found similar increases in arachidonate/linoelic acid ratio and in GSH levels without bearing on peptido-leukotriene levels. The authors also found the influence of a low calcium and methionine diet on Pb-induced serum fatty acid profiles (Knowles and Donaldson, 1990).

Chronic sublethal exposure (5 ppm Pb-nitrate for 30 days) has been found to alter liver lipid profiles in blood and liver tissue of the fresh water fish *Anabas testudineus* (Tulasi et al., 1992). These authors reported significant increases in liver total lipids, cholesterol, and free fatty acids. Tandon et al. (1994b) reported that iron deficiency enhanced the accumulation of Pb in liver and kidney and also increased liver calcium levels. Induced expression of metallothionein (MT) in renal and intestine was also observed in iron deficiency. Han et al. (1996) investigated the effect of Pb burden on weight loss using an energy restriction diet regimen on rats with prior Pb exposure. The authors reported that rats on a substantial weight loss regimen (40% of normal calories) exhibited a significant increase in the quantity and concentration of liver Pb and a decrease in the concentration of other metals (e.g., Ca, Cu, Mg, Zn). The authors concluded that

weight loss can increase the liver concentration of Pb, even in the absence of continued exposure. Combined exposure to Pb (70 mg/kg) and Cd (20 mg/kg) in Buffalo rats for 7 weeks was found to alter liver levels of Zn and Cu, with less accumulation of Pb and Cd, compared to individuals exposure to either Pb or Cd alone (Skoczynska et al., 1993). These authors also reported that a combined exposure regimen interfered with serum lipid profiles (Skoczynska and Smolik, 1994).

Liu et al. (1997) utilized rat primary hepatocyte cultures to explore the protective effect of Zn-induced expression of metallothionein (MT) in Pb toxicity. These authors found that, in the control cells without prior Zn exposure, most of the Pb was found bound to high-molecular weight proteins in the cytosol, while in the Zn pretreated cells, a majority of Pb bound to MT, indicating a MT-mediated protection against Pb toxicity to hepatocytes. More details about these and related studies are summarized in Table AX5-10.2.

5.10.1.3 Effects of Lead Exposure on Hepatic Cholesterol Metabolism

Lead-nitrate-induced hyperplasia or liver cell proliferation involves simultaneous increase in both liver and serum total cholesterol levels. Recent studies have reported various molecular events associated with this process. Induction of gene expression for CYP51 (Lanosterol 14α-demethylase), an essential enzyme for cholesterol biosynthesis, was reported in Pb-nitrate-induced liver hyperplasia, although other cytochrome P450 enzymes involved in drug metabolism have been reported as being suppressed, as discussed in earlier sections. This gene has various regulatory elements and its constitutive expression in liver is mediated by sterol regulatory element (SRE) and by the SRE binding proteins-1a, 2, and 1c. Kojima et al. (2002) reported that Pb-nitrate induced the expression of CYP51 in the livers of both immature (4-week-old) and mature (7-week-old) rats and that this induction appeared to be mediated by the upregulation of SRE binding protein-2. However, this increased synthesis of cholesterol observed in rat liver was not mediated by endogenous feedback regulation by sterols, as no decrease in serum total cholesterol was observed. To understand the molecular mechanisms involved in the Pb-nitrate-mediated development of hepatic hypercholesterolemia, Kojima et al. (2004) investigated the expression of various enzymes involved in cholesterol homeostasis, including some of the associated transcription factors in male rats exposed to Pb-nitrate (100 μmol/kg body wt). The authors reported that Pb-nitrate exposure caused a significant

increase in liver and serum total cholesterol levels at 3-72 h and 12-72 h, respectively. The enzymes involved in cholesterol biosynthesis viz. (i.e., 3-hydroxy-3methyglutaryl-CoA reductase, farnesyl diphosphate synthase, squalene synthase, CYP51) were all activated (3-24 h), while the enzymes involved in cholesterol catabolism such as 7α-hydroxylase were remarkably suppressed 3-72 h. Figure 5-10.1 shows the involvement of Pb at various stages of the cholesterol synthesis pathway. The induction of the cytokines interleukin-1α and TNF-α in rat liver prior to the induction of the genes for these synthesis enzymes suggested that Pb-nitrate-induced cholesterol synthesis is independent of sterol homeostasis regulation. Following gestational and lactational exposure to Pb-acetate (0.05 mg/kg body wt), Pillai and Gupta (2005) reported that the activities of the hepatic steroid metabolizing enzyme 17-β-hydroxy steroid reductase, UDP glucouronyl transferase, and CYP450 levels decreased in rat pups on PND21.

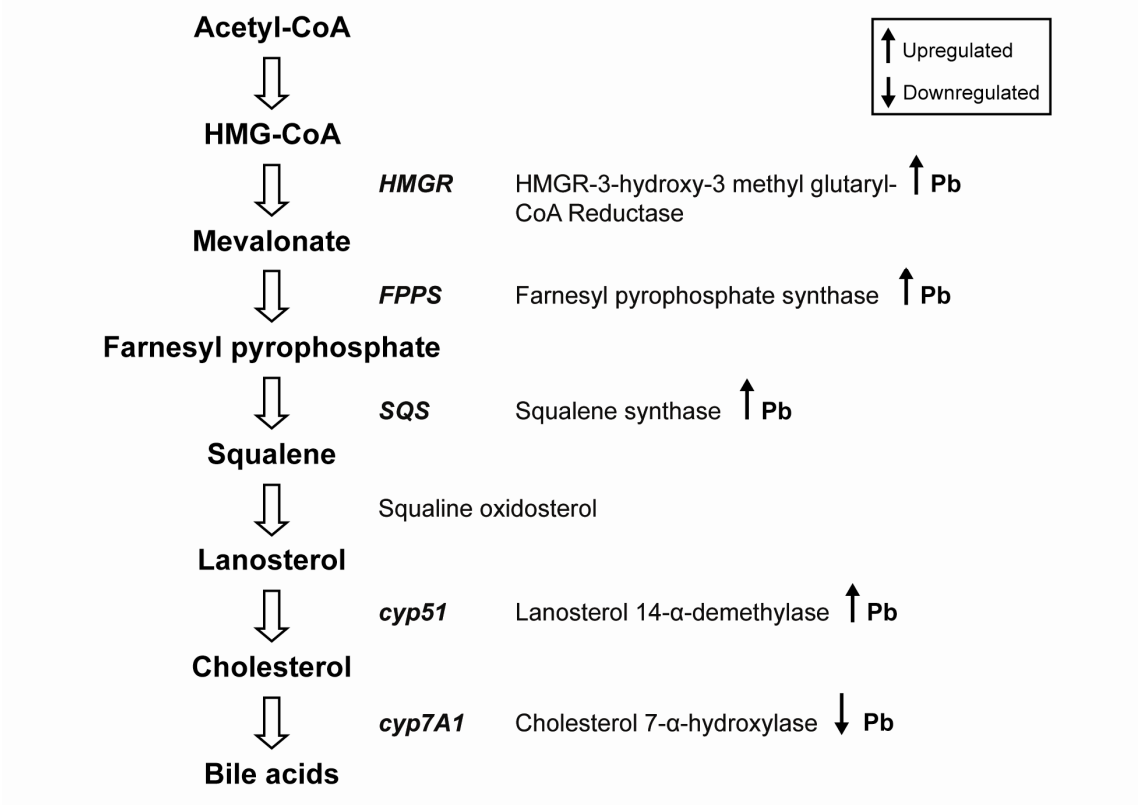

Figure 5-10.1. Flow diagram indicating the Pb effects on the cholesterol synthesis pathway.

Alterations in the hepatic system of neonates and pups (at PND12 and PND21) after gestational and lactational exposure to Pb-acetate (300 mg/L) have been reported by Corpas et al. (2002). The authors found significant reductions in the liver weight of pups and in hepatic glycogen that correlated with increased blood glucose levels. The authors also reported reductions in liver protein, lipid levels, and alkaline and acid phosphatase activities but did not find any gross structural alterations in liver tissue. These and other studies are summarized in Table AX5-10.3.

5.10.1.4 Effect of Chelation Therapy on Lead-Induced Hepatic Oxidative Stress

Although several mechanisms have been proposed to explain Pb toxicity, no mechanism has been defined explicitly. Recent literature on Pb toxicity suggests oxidative stress as one of the important mechanisms of toxic effects of Pb in liver, kidneys, brain, and other organs.

Schematic representation of the various mechanisms by which Pb induces lipid peroxidation is shown Figure 5-10.2. Lead toxicity to the liver has been found to be associated with significant accumulation of Pb in the liver. This results in the accentuation of lipid peroxidation with concomitant inhibition of antioxidant enzymes (i.e., SOD, catalase, GSH peroxidase, GSH reductase) and a simultaneous increase in GSSG with a reduction in GSH/GSSG ratio (Sandhir and Gill, 1995; Aykin-Burns et al., 2003). However, Furono et al. (1996) studied the potential of various redox-active metals to induce LPO in normal and alpha-linolenic acid-loaded rat hepatocytes and suggested that Pb ions were not capable of inducing lipid peroxidation in such hepatocytes.

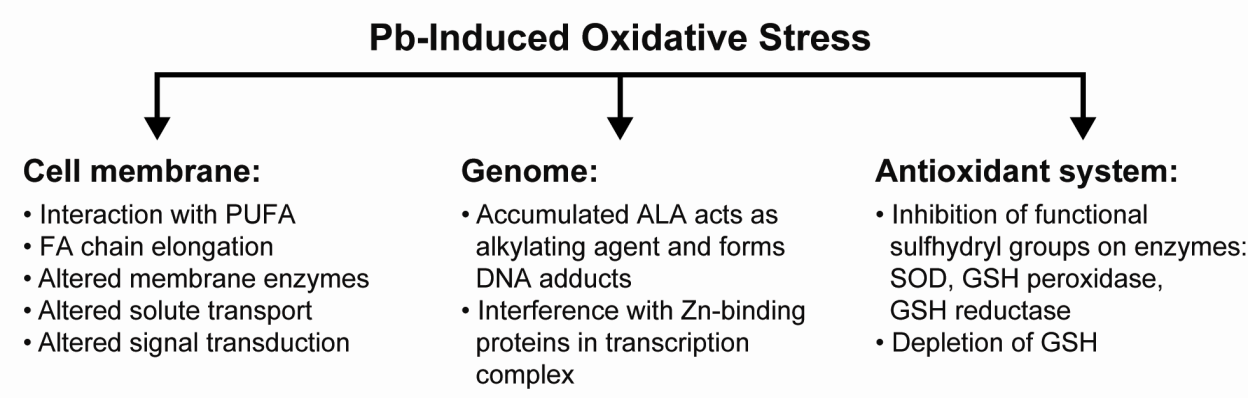

Figure 5-10.2. Schematic diagram illustrating the mode of Pb-induced lipid peroxidation.

The currently approved clinical intervention method is to give chelating agents that form an insoluble complex with Pb and remove the same from Pb-burdened tissues. The efficacy of various chelating agents and antioxidants studied in experimental animals on Pb induced liver toxicity is discussed below.

Chelation therapy with mono-3-methylbutane-1-yl (monoisomyl) ester of meso-2,3-dimercaptosuccinic acid (Mi-DMSA) and meso-DMSA (meso-2,3-dimercaptosuccinic acid) was found to offer no protection to suckling rat pups as measured by liver Pb levels (Cory-Slechta, 1988; Blanusa et al., 1995; Pappas et al., 1995; Smith et al., 2000). Similar studies by Kostial et al. (1999), using various isoforms of DMSA, EDTA, and combined therapy, did not find a chelator-mediated reduction in liver Pb levels except for meso-DMSA (0.5 mmol/kg), which caused significant reduction in Pb levels in kidney and brain. On the other hand, it decreased liver zinc and copper levels. The authors concluded that combined therapy may not be the best choice at this age, because infants are more sensitive to trace metal deficiency. Flora and Seth (1999) investigated the protective role of S-adenosyl-L-methionine (SAM) on acute Pb and Pb + thanol-induced hepatic toxicity in mice by monitoring hepatic GSH and MDA levels. The authors concluded that bioaccumulation of Pb in liver in both Pb- and Pb + ethanol-exposed groups were significantly decreased by SAM.

To identify the efficacy of chelation therapy (mono or combined therapy) for acute Pb poisoning in infants, Kostial et al. (1999) utilized suckling rat pups and monitored tissue Pb levels. Monotherapy of either EDTA (0.3 mmol/kg), meso-DMSA (0.5 mmol/kg), rac-DMSA (racemic-2,3-meso-2,3-dimercaptosuccinic acid, 0.5 mmol/kg), or a combined therapy of EDTA + meso-DMSA, EDTA + rac-DMSO indicated differential effects on liver tissue Pb and other trace metal levels. The authors concluded that meso-DMSA was the more potent therapy for acute Pb poisoning in infants and suggested that combined therapy may not be the best choice, as at this age the infants are more sensitive to trace metal deficiency.

Supplementation with sodium molybdate (1 mg/kg body wt) during the course of Pb exposure (0.1% Pb-acetate in water for 4 weeks) was found to provide significant protection from the uptake of Pb by blood, liver and kidneys and also from hepatic LPO (Flora et al., 1993). Similarly, supplementation with antioxidants and vitamins were explored to reduce the toxic effects of Pb on liver function and activity. Oral supplementation of vitamin C (100 mg/kg for 3 days) has been reported to provide significant protection against Pb-induced declines in liver

heme synthesis, drug metabolism, tissue thiols, and vitamin C levels along with reduction in liver Pb levels (Vij et al., 1998). Similarly, simultaneous administration of vitamin E (5 mg/kg body wt) was reported to confer protection against Pb-induced decline in hepatic type-1 iodothyronine 5'-monodeiodinase activity, inhibition of SOD and catalase activities, and increased lipid peroxidation (Chaurasia and Kar, 1997). Studies by Tandon et al. (1997) also suggested that Pb and Pb + ethanol-induced biochemical changes in mouse liver can be prevented by the simultaneous administration of lysine and zinc. This regimen was also reported to prevent the Pb-induced depletion of endogenous calcium and magnesium in liver. Two well-known antioxidants, N-acetylcysteine and lipoic acid, have been reported to reduce Pb-induced oxidative stress (OS) both in vitro in Chinese hamster ovary cells and in vivo in F344 rats (Ercal et al., 1996; Gurer et al., 1998, 1999b). The same group also investigated the protective effects of another antioxidant, taurine, against Pb-induced OS in the same systems both in vitro and in vivo. These authors reported that taurine was effective by increasing cellular GSH while simultaneously reducing malondiablehde (MAD) and catalase activity levels, offering protection against Pb-acetate-induced OS, without decreasing the liver or blood Pb levels (Gurer et al., 2001). Patra et al. (2001) studied the ameliorative effects of antioxidants (i.e., ascorbic acid, vitamin E, L-methionine) alone and vitamin E + EDTA on Pb-induced OS in liver, kidney, and brain tissues of rats exposed to Pb-acetate (1 mg/kg body wt, 4 weeks) and found that all the antioxidants used conferred protection against OS without a significant decline in tissue Pb burden. The level of protection conferred exhibited tissue-specific differences. L-Methionine was also found to offer similar protection in mice exposed to Pb (Xie et al., 2003). Othman and El Missiry (1998) reported that administration of selenium (sodium selinite, 10 μM/kg body wt) prior to Pb-acetate (100 μM/kg body wt) produced pronounced prophylactic action against Pb-induced LPO in liver and kidney of male albino rats.

In earlier combination chelation therapy using thiamine and Ca^{2+}-EDTA, Kim et al. (1992) reported that regardless of the route of exposure, reduction in liver tissue retention of ^{203}Pb occurred, while thiamine alone reduced only the Pb content of kidney. Recent studies used a combination of chelators with antioxidants to reduce Pb-induced OS in liver and other tissues (i.e., kidney and brain). α-Lipoic acid, meso-DMSA, and their combination was found to reduce OS by increasing hepatic GSH levels and reducing GSSG and thiobarbituric acid reactive substances (Pande and Flora, 2002). The same group also studied the protective effect of the

combination of ascorbic acid, vitamin E, meso-DMSA, and miADMSA and found a significant reduction in hepatic OS by the combination therapy of ascorbic acid and thiol chelators (i.e., DMSA, miADMSA) in rat. The combination therapy also produced similar reduction in renal OS (Flora et al., 2003). Studies reported by Varnai et al., (2003) suggested that ascorbic acid supplementation did not improve the efficiency of meso-DMSA in reducing Pb-induced OS in suckling rats. On the other hand, combined treatment of ascorbic acid (1 mg/100 g body wt) and silymarian (1 mg/100 g body wt) has been reported (Shalan et al., 2005) to cause marked improvement of the biochemical, molecular and histopathological changes caused by Pb-acetate (500 mg/kg body wt). Similarly, combined treatment with lipoic acid + DMSA has been found to completely ameliorate Pb-acetate-induced oxidative damage. However, either lipoic acid or DMSA alone conferred partial protection against Pb-induced hepatic damage (Sivaprasad et al., 2004). These and related studies are summarized in Table AX5-10.4.

5.10.1.5 Lead-Induced Liver Hyperplasia: Mediators and Molecular Mechanisms

The biochemical and molecular events associated with Pb-induced hyperplasia has been accumulating in the scientific literature. Lead-nitrate, a known mitogen, is also considered to be a carcinogen that induces liver cell proliferation in rats without any accompanying liver cell necrosis. It has been recognized that this proliferation is a transient process and that apoptosis plays a major role in the regression of Pb-nitrate-induced hepatic hyperplasia (Nakajima et al., 1995). Columbano et al. (1996) studied the cell proliferation and regression phases by apoptosis in Wistar male rat liver by monitoring the incorporation of tritiated thymidine as a marker for increased DNA synthesis. These studies demonstrated the production of Pb-induced proliferation 3 days after a single injection of Pb-nitrate with complete regression of hyperplasia seen after 15 days. The authors suggested that the apoptosis process observed in the regression phase also involved newly initiated hepatocytes. On the other hand, Dini et al. (1999) reported the regressive or involutive phase as beginning 5 days post single injection of Pb-nitrate. Apostoli et al. (2000) evaluated the proliferative effects of various Pb salts (i.e., Pb-acetate, Pb-chloride, Pb-monoxide, Pb-sulfate) using liver-derived REL cells. These authors reported that all the Pb compounds tested showed dose- and time-dependent effects on the proliferation of REL cells. Unlike other tumor promoters, Pb compounds did not exhibit effects on cell junctional coupling. Liver hyperplasia induced by Pb-nitrate has been shown to demonstrate

sexual dimorphism in all phases of the proliferation as well as in apoptosis (Tessitore et al., 1995). Biochemical changes associated with liver hyperplasia in the intermediary metabolic pathways were discussed in earlier sections of this chapter; the present discussion focuses on other molecular characteristics of this process. As the numerous molecular networks involved in both the proliferation and apoptosis processes have many common mediators and pathways, it is very difficult to provide a discussion without an overlap.

DNA hypomethylation has been recognized to play a major role in the proliferation of cells in regenerating and in hepatic pre-malignant lesions when compared to normal non-dividing liver cells. A single dose of Pb-nitrate (75 μM/kg body wt) has been found to cause extensive hypomethylation in rat liver (Kanduc et al., 1991). Additional investigations from the same group reported that this hypomethylation status of liver DNA by Pb-nitrate changed significantly with age and exhibited liver cell specificity (Kanduc and Prisco, 1992).

Investigations of cell cycle-dependent expression of proto-oncogenes in Pb-nitrate (10 μM/100 g body wt)-induced liver cell proliferation by Coni et al. (1989) showed that peak DNA synthesis occurred at 36 h after a single injection of Pb-nitrate. In addition to DNA synthesis, induced expression of c-fos, c-myc, and c-Ha-ras oncogenes was also observed in rat liver tissue. Additional studies by the same group reported that Pb-nitrate-induced liver hyperplasia involved an increased expression of c-jun in the absence of c-fos expression (Coni et al., 1993). The induced expression of c-myc persisted up to 40 h post Pb-nitrate exposure. Pb-nitrate-induced liver proliferation and DNA synthesis, as monitored by 5-bromo-2-deoxyuridine immunohistochemistry, lead to DNA labeling in a few hepatocytes (Rijhsinghani et al., 1993). The observed DNA synthesis appeared to be due to the increased activity and expression of DNA polymerase-α observed at 8 h postexposure to a single injection of Pb-nitrate (Menegazzi et al., 1992). Along with DNA synthesis, poly (ADP-ribose) polymerase was also induced by Pb-nitrate (Menegazzi et al., 1990). Differential activation of various PKC isoforms, downregulation of PKC-α, and marked activation of PKC-ϵ in Pb-nitrate-mediated liver hyperplasia suggested the involvement of these PKC enzymes in DNA synthesis and related signal transduction pathways (Tessitore et al., 1994; Liu et al., 1997).

Coni et al. (1992) reported the proliferation of normal and pre-neoplastic hepatic cells treated with the plasma derived from male Wistar rats treated with a single injection of Pb-nitrate; this was the first report on the secretion of biological cell proliferation signals in the liver

after Pb-nitrate treatment. These authors reported that DNA synthesis was detected as early as 30 min and persisted up to 5 days after Pb-nitrate exposure. This observation has opened up the inquiry into the involvement of various growth factors and other biological mediators in hepatic hyperplasia. Shinozuka et al. (1994) investigated the expression of various growth factors (i.e., hepatocyte growth factor, TGF-α, TGF-β) in rat liver after a single injection of Pb-nitrate (100 μM/kg body wt) and reported the involvement of these growth factors in liver cell proliferation. Additional studies by this group to observe LPS sensitivity in rats given Pb nitrate reported that animals given a single injection of LPS up to 100 μg survived, whereas in the presence of Pb-nitrate, they tolerated only 6 μg of LPS, indicating that Pb-nitrate may sensitize the animals for LPS toxicity.

Earlier studies by Honchel et al. (1991) reported that coexposure of rats to Pb-acetate (15 mg/kg) and LPS or TNF showed markedly increased serum levels for various liver injury parameters. They concluded that Pb may potentiate liver toxicity by LPS via a TNF-mediated pathway. The role of TNF-α in Pb-nitrate-induced liver cell proliferation was further investigated by (Ledda-Columbano et al., 1994) who demonstrated the inhibition of Pb-nitrate-induced cell proliferation by pretreatment with dexamethasone, an inhibitor of TNF-α expression. Additional studies by the same group evaluated the liver cell specificity in Pb-nitrate-induced cell proliferation (Shinozuka et al., 1996). They monitored the incorporation of 5-bromo-2-deoxyuridine by immunohistochemical analysis on rat liver as induced by Pb-nitrate and TNF-α and observed 5-bromo-2-deoxyuridine incorporation in hepatocytes and non-parenchymal cells (i.e., Kupffer cells, endothelial cells, periportal nondescript cells), confirming that Pb-induced liver cell proliferation was mediated by TNF-α. Kubo et al. (1996) used various TNF-α inhibitors to further confirm the role of TNF-α in Pb-nitrate-induced hepatocyte proliferation. Menegazzi et al. (1997) reported that Pb-nitrate induced proliferation involved the induction of iNOS along with TNF-α and that appeared to be mediated by a strong, prolonged activation of NFκB but not activator protein-1 (AP-1). Nemoto et al. (2000) investigated the potential role of neurotrophins and their receptors in Pb-nitrate-induced hepatic hyperplasia. The expression profile of TNF-α, neurotrophins (i.e., nerve growth factor, brain-derived neurotrophic factor neurotrophin-3 and (their receptors), tyrosine kinase receptor (Trk) and neurotrophin receptor (p75NTR) were investigated in liver tissue after a single injection of Pb-nitrate (100 μM/kg body wt). The Pb-nitrate induced increased expression of TNF-α preceded the

expression of the neurotrophins and their receptors. Based on these results, the author's suggested that neurotrophins and neurotrophin receptors are involved in mediating mitogenic signals related to hepatic hyperplasia.

The regression phase of Pb-induced liver hyperplasia appears to be mediated by OS. As discussed earlier, this process involves LPO and other cytokine mediators, including TNF-α. Sieg and Billings (1997) reported that Pb potentiated cytokine-induced OS, producing a significant decline in intracellular ATP concentration in mouse hepatocyte culture studies. The authors suggested that cytotoxic interaction between Pb and cytokines (e.g., TNF-α and IFN) may be mediated by oxidative DNA damage resulting from OS. The potential role OS along with TNF-α has been implicated in the apoptosis of hepatocytes by Milosevic and Maier (2000). Using freshly isolated cultures of hepatocytes and Kupffer cells and their co-culture system exposed to Pb-acetate (2-50 μM) and LPS (0.1-1000 ng/mL), the authors reported that, in the co-culture system, the Pb-LPS-induced release of TNF-α from the Kupffer cells, increased nitric oxide levels by 6-fold and downregulated the acute phase protein, albumin, in hepatocytes. From these observations the authors concluded that Pb-induced Kupffer cell-derived signals promoted the toxicity of Pb in hepatocytes, resulting in hepatocyte death by proteolysis. The importance of the Kupffer cells role in Pb-nitrate-induced heptatocyte apoptosis was further demonstrated (Pagliara et al., 2003a,b). These authors reported that in vivo hepatic apoptosis including oxidative response induced by Pb-nitrate, was prevented by pretreatment with gadolinium chloride, a Kupffer cell toxicant that specifically suppresses Kupffer cell activity. When treated hepatocytes were exposed in vitro to Pb-nitrate, hepatocyte apoptosis was not observed. On the other hand, hepatocyte apoptosis was evident when the hepatocytes were incubated with culture medium derived from Kupffer cells that had been exposed to Pb-nitrate. Based on these studies, the authors concluded that heptocyte apoptosis was potentiated by soluble factors secreted by Pb-exposed Kupffer cells. The role of activated Kupffer cells, macrophages, and TNF-α in chemical-induced hepatotoxicity is presented schematically in Figure 5-10.3.

Dini et al. (1993) investigated the expression of asialoglycoprotein receptors on the surface of hepatocytes and galactose-specific receptors of non-parenchymal cells during the apoptic phase of Pb-induced hepatic hyperplasia. A significant increase in asialoglycoprotein

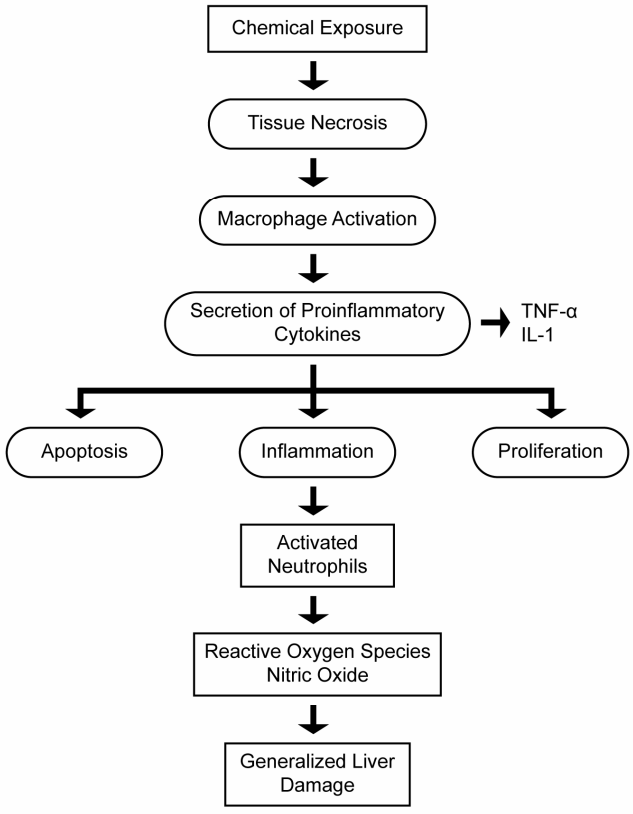

Figure 5-10.3. Hypothesis of chemical-induced liver injury generated primarily on the basis of different types of inhibitors.

receptor expression in hepatocytes coincided with massive apoptosis. Later studies from this group demonstrated that sinusoidal liver cells predominantly phagocytosed the Pb-nitrate-induced apoptic hepatic cells and concluded that this process appeared to be mediated by the cell surface carbohydrate receptors (i.e., mannose and galactose receptors) (Ruzittu et al., 1999). Pretreatment of rats with gadolinium chloride, a kupffer cell toxicant, was also found to abolish the altered expression of galactose receptors (Pagliara et al., 2003b).

The role of glucocorticoid-mediated signal transduction in the hepatotoxicity of Pb was evaluated by Heiman and Tonner (1995), using H4-IIE-C3 hepatoma cells (HTC). Acute exposure of cells to Pb (300 nM^{-1} or 10 μM) was found to inhibit processes involved in glucocorticoid-mediated enzyme induction (e.g., tyrosine aminotransferase activity) in a dose-dependent manner both at the transcriptional and translational level, without altering glucocorticoid receptor binding characteristics. Tonner and Heiman (1997) also reported Pb-induced hepatotoxicity by glucocorticoid-mediated signaling and its involvement in the

interference with calcium-mediated events as well as the differential modulation and translocation of protein kinase isoforms α and β into the nucleus. More information on these and other related studies is summarized in Table AX5-10.5.

5.10.1.6 Effects of Lead on Liver Heme Synthesis

Effects of Pb on heme metabolism have been extensively investigated in major target tissues such as liver and erythrocytes. Section 5.2 described Pb effects on heme synthesis, with particular relevance to erythrocytes. The effects of Pb on heme synthesis in the liver and the role of chelation therapy in this process are discussed in this section.

Fifteen percent of heme is produced in the liver. Heme metabolism in the liver is an essential component of various cytochrome P450s that participate in cellular redox reactions and xenobiotic detoxification pathways in the liver tissue and, hence, heme plays a vital role in liver function (Jover et al., 1996). Due to the important and critical role of heme in liver function, Pb-induced effects on hepatic heme metabolism are discussed below.

Initial studies on the effects of Pb-nitrate on hepatic heme biosynthesis were reported by Lake and Gerschenson (1978) using the rat liver cell line (RLC-GAI). The effects of various organic metal compounds on ALAD activity have been studied by Bondy (1986). The authors reported that triethyl Pb-chloride has the same potency as Pb-nitrate in inhibiting ALAD both in vitro and in vivo, with liver and blood ALAD exhibiting similar sensitivities to Pb compounds. By measuring the conversion of ALA into heme, these authors showed that heme biosynthesis was inhibited by Pb in a dose dependent manner. Using a lipophilic complex of Pb-acetate + DTC to increase the cellular uptake of Pb, Osksarsson et al. (1989) demonstrated the inhibition of ALAD activity in primary rat hepatocytes cultures. Lead-acetate has been reported to inhibit ALAD activity in rabbit liver tissue without any effect on delta-aminolevulinic acid synthase (ALA-synthase) activity (Zereba and Chemielnicka, 1992). Exposure to Pb (500 ppm) in drinking water did not inhibit hepatic ALA-synthase, but did inhibit ALA-dehydratase activity in mice (Tomokuni et al., 1991). Exposure to Pb-acetate (20 mg/kg body wt for 3 days) has been reported to decrease hepatic ALAD and uroporphyrinogen activity (Satija and Vij, 1995). These authors also reported that IP injection of zinc (5 mg/kg body wt for 3 days) conferred protection against Pb-acetate effects in liver tissue.

Effects of Pb on hepatic porphyrins, intermediate metabolites of heme metabolism, were investigated by few researchers. Quntanilla-Vega et al. (1995) reported that 3T3-hepatocyte cultures, when incubated with a micromolar concentration of Pb-acetate increased cellular porphyrin content and excretion. This increased porphyrin production may have been due to an accumulation of protoporphyrin and coproporphyrin, as in coproporphyrinuria, a well-characterized sign of Pb intoxication (Ichiba and Tomokuni, 1987; Zereba and Chemielnicka, 1992). Dietary supplementation of selenium and monensin increased Pb-induced accumulation of prophyrins in chicken liver (Khan and Szarek, 1994). Species-specific differences in the effects of Pb on protoporphyrins were reported by Jacobs et al. (1998). These authors investigated the effect of Pb on zinc protoporphyrin synthesis in cultured chick and rat hepatocytes and observed decreased levels of protoporphyrin in rat hepatocytes, but no effect on chick hepatocytes. Santos et al. (1999) also reported Pb-induced derangements (including porphyrin metabolism) in rat liver heme metabolism, but these effects were far less severe than those observed in erythrocytes. Their investigations on the effect of chronic alcoholism on Pb effects in hepatic heme metabolism suggested no potentiation by alcohol.

Transferrin (TF) is the major iron-transport protein in serum and other biological fluids. Transferrin can also has the capacity to transport other metals. Lead was found to inhibit TF endocytosis and transport of iron across the cell membrane of rabbit reticulocytes (Qian and Morgan, 1990). The effect of Pb on TF gene expression was investigated by Adrian et al. (1993) using a transgenic mouse with the human TF gene. They found that Pb suppressed the expression of TF transgene in mouse liver at the transcriptional level; however, the same dose of Pb did not inhibit mouse endogenous hepatic TF gene expression. Lead exposure was also found to inhibit recombinant TF expression in human hepatoma hepG2 cells. Other studies by the same group found that Pb exposure suppressed the expression of endogenous TF in HepG2 cells (Barnum-Huckins et al., 1997). These authors further suggested that Pb effects on hepatic TF levels may also interfere with iron metabolism in humans. (See Annex Table AX5-10.6 for more information on these and related studies.)

5.10.1.7 Summary

Extensive in vivo and in vitro experimental evidence has accumulated over the past 20 years and increased our understanding of the potential toxic effects of Pb in the hepatic

system. These studies ranged from simple biochemical studies to molecular characterizations of the induction of drug-metabolizing enzymes, liver hyperplasia, and the protective effects of chelation therapy.

- Rat liver microsomal cytochrome P-450 levels were found to decrease with a single dose exposure of Pb nitrate. Inhibition of both constitutive and induced expression of microsomal P450 A1 and A2 activity occurred. Simultaneous induction of the activities of phase II drug metabolizing enzymes with decreased phase I enzymes with single exposure to Pb nitrate suggests biochemical properties similar to hepatic nodules.
- Newer studies examined the induction of GST-P at both transcriptional and translational levels using in vitro systems and indicated a role for Pb-nitrate and Pb-acetate in the induction process. On the other hand, triethyl Pb compounds have been found to suppress the activity of various GST isoforms.
- Studies on Pb-induced liver hyperplasia demonstrated de novo synthesis of cholesterol, alterations in the gluconeogenic mechanism, as well as DNA hypomethylation and subsequent changes in the expression of protooncogenes.
- Lead-induced alterations in cholesterol metabolism appear to be mediated by the induction of several enzymes related to cholesterol metabolism and the decrease of 7 α-hydroxylase, a cholesterol catabolizing enzyme. This regulation of cholesterol homeostasis is modulated by changes in cytokine expression and related signaling.
- Studies using an inhibitor to block TNF-α have clearly demonstrated TNF-α as one of the major mitogenic signals that mediate Pb-nitrate-induced liver hyperplasia. Lead-induced hyperplasia also appears to be modulated by neurotrophins and their receptors.
- In vitro co-culture systems with Kupffer cells and hepatocytes suggested liver cell apoptosis is mediated by Kupffer cell-derived signals and Pb-induced oxidative stress.
- Newer experimental evidence suggests that Pb-induced alterations in liver heme metabolism involves perturbations in ALAD activity, and porphyrin metabolism, alterations in Transferrin gene expression, and associated changes in iron metabolism.
- Limited experimental evidence on the role of weight loss on liver Pb burden in exposed animals indicate that liver Pb content increases even in the absence of prolonged continued exposure.
- Extensive scientific evidence has accumulated over these two decades on the role of chelation therapy, both individual and combined. Studies using a combination of therapy regimens with chelators such as DMSA, Mi-DMSA, or DMSA+ EDTA did not prove beneficial in ameliorating the Pb-induced oxidative stress in infant/neonatal rats as the combination therapy in young rats resulted in essential mineral deficiencies.
- Therapeutic intervention with S-adenosyl-L-methionine, L-acetyl cysteine, lipoic acid, and vitamin E conferred protection against Pb accumulation in the liver and Pb-induced lipid peroxidation. Intervention with ascorbic acid, on the other hand, has been found to confer protection against Pb-induced decrease in hepatic heme synthesis.

5.10.2 Gastrointestinal System and Lead Absorption

Lead enters the body by many routes, but primarily via the GI tract. The intestinal epithelium serves as one of the body's primary interfaces with the outside world. The transporting epithelia in the small intestine are characterized by layers of anatomically and biochemically polarized cells that are connected to each other by tight junctions and resting on a basement membrane. Classically, the intestinal epithelium is thought of primarily as a barrier, but it also is a highly reactive barrier. Even modest perturbations in its functions may lead to diarrhea, constipation, malnutrition, dehydration, and infectious diseases (i.e., ulcerative colitis, collectively referred as chronic intestinal inflammatory diseases) (Gewirtz et al., 2002). Abdominal colic and constipation are symptoms of Pb poisoning, but its mechanism is not fully understood. Studies have been carried out in the past decade to increase our understanding of the fundamental mechanism(s) in order to extrapolate the experimental observations to human health effects.

The intestinal absorption of Pb is influenced by a variety of factors, including the chemical and physical forms of the element, age at intake, and various nutritional factors. Gastrointestinal absorption of Pb is thought to occur primarily in the duodenum. In the isolated rat intestine, absorption, and, in particular, serosal Pb transfer activity (net transfer of Pb from the small intestine lumen across the epithelium and into the serosal space) is highest in the duodenum. The mechanisms of absorption may involve active transport and/or diffusion through the intestinal epithelial cells. Both saturable and non-saturable pathways of absorption have been inferred from the studies in different animal models, although the understanding of the former is slightly more robust (Diamond et al., 1998).

Transport of Pb as a complex with proteins via endocytosis or as a complex with amino acids are postulated as possible mechanisms. Direct evidence for transport of an organic Pb complex has not been provided, but it seems possible.

In the cell, Pb interacts with a variety of intracellular ligands, including calcium-binding proteins and high-affinity Pb-binding proteins. Transfer across the cell or basolateral membrane (or both) involves a mechanism(s) that may be sensitive to vitamin D and iron status. Alternate transport mechanisms via a Ca^{2+}-Na^{+} exchanger, independent of regulation by vitamin D, are also possible.

5.10.2.1 Lead and In vitro Cytotoxicity in Intestinal Cells

In vitro cytotoxicity of metal salts for 48 h was determined in the intestinal epithelial cell line I-407 by Keogh et al. (1994). The investigations identified rank order cytotoxicity in terms of LC_{50} values: $HgCl_2$ (32 μM)>$CdCl_2$ (53 μM) > $CuCl_2$ (156 μM) > Ti_2SO_4 (377 μM) > Pb $(NO_3)_2$ (1.99 mM). Further studies using a noncytotoxic concentration of butathione sulphoxamine pretreatment for GSH depletion revealed that the cytotoxicity of Pb was unaffected by GSH depletion (see Table AX5-10.7).

5.10.2.2 Alterations in Intestinal Physiology and Ultrastructure

Karmakar et al. (1986) investigated the pathologic alterations that occur in the intestine, liver, and kidney of Pb-intoxicated rats upon short-term exposure to sublethal doses of Pb (44 mg/Kg body wt) and reported degeneration of intestinal mucosal epithelium leading to potential malabsorption.

The effect of low-concentration Pb-acetate (0.1%) on the jejunal ultrastructure was studied by Tomczok et al. (1988) in young male rats. The studies revealed that the villi of jejunum of rats exposed to Pb for 30 days had a rough appearance on the surface, which could be associated with a distortion of glycocalyx layer. Areas of extensive degenerative lesions were also observed on the surface of most villi on the 60th day of exposure. All intestinal epithelial cells exhibited various degrees of glycocalyx disturbance, indicating that pronounced toxic effects of Pb were related to modifications of the biochemical properties of the surface coat of the cells. These authors also reported the appearance of goblet cells and of Pb deposition along the goblet cell membrane in blocks of tissue along the border between duodenum and jejunum. Continued treatment up to 60 days resulted in mucus droplets in the cytoplasm of goblet cells, along with deposition of silver salts indicative of Pb in these cells. These results demonstrated the significance of goblet cells in Pb detoxification.

In another study on the ultrastructure of rat jejunum exposed to Pb-acetate (100 mg/kg body wt/day), Tomczok et al. (1991) found that 30-day treatment resulted in numerous small, rough-membraned vesicles and dilated golgi complexes in the cytoplasm. Continued treatment for 60 days resulted in vacuolated cytoplasm associated with the golgi complexes, rough-membraned vesicles, and dilated cisternae. Also, the surface of the intestinal epithelial cell microvilli showed evidence of Pb deposition, as evidenced by Timm sulfide silver reaction sites.

5.10.2.3 Intestinal Uptake and Transport

Infants are a particularly susceptible population for Pb toxicity, possibly due to the immaturity of the digestive tract, feeding pattern, or source of Pb. To investigate these aspects, Henning's group (Beach and Henning, 1988; Henning and Cooper, 1988) carried out a series of experiments using suckling rat pups and reported that Pb in rat and bovine milk and infant milk formula was primarily associated with casein micelles. Casein-bound Pb may be the most common form of Pb presented to the small intestine (Beach and Henning, 1988). Other studies by this group investigated potential differences in the mechanisms when Pb was presented in ionic or milk-bound form, using ^{203}Pb as a tracer. These studies clearly showed that when ^{203}Pb was administered intragastrically as a soluble salt, it was primarily accumulated in the duodenum, regardless of dose or vehicle used. In contrast, substantial accumulation of ^{203}Pb was found in the ileal tissue following Pb administration in milk. These studies clearly indicated strikingly different patterns in the intestinal accumulation of ionic and milk-bound Pb and suggest a greater toxicity for Pb in drinking water compared to Pb ingestion via milk (Henning and Cooper, 1988).

Dekaney et al. (1997) investigated the uptake and transport of Pb using intestinal epithelial cells (IEC-6). The authors observed that Pb accumulation in Pb-exposed (5-10 μM) IEC-6 cells was time- and dose-dependent up to 1 h and that reduction of the incubation temperature significantly reduced the total cellular Pb content of IEC-6 cells. Simultaneous exposure to Zn resulted in decreased cellular Pb content compared to cells exposed to Pb only. Exposure of cells to ouabin or sodium azide has been found to increase Pb accumulation in the cells compared to cells treated with Pb (5 μM) alone. These studies clearly demonstrate that Pb transport in IEC-6 cells is time- and temperature-dependent, involves the presence of sulfydryl groups, and competes with the uptake of Zn.

Lead speciation and transport across intestinal epithelium in artificial human digestive fluid (chyme), both in vivo and in vitro, in Caco-2 cells were evaluated by Oomen et al. (2003). In vivo studies indicated that in chyme, Pb-phosphate and Pb-bile complexes are important fractions. The metal ions dissociated from these complexes can subsequently be transported across the intestinal epithelium or they may traverse the intestinal membrane. In vitro studies, on the transport of bioaccessible Pb across the intestinal epithelium in Caco-2 cells exposed to diluted artificial chyme for 24 h, indicated that 3% of the Pb was transported across the cell

monolayer. Lead associated with cells in a linear relationship to the total amount of Pb in the system. Bile levels were not found to affect the fraction of Pb associated with the cells. The free Pb^{2+} concentration in chyme was negligible. Extrapolating these results to the in vivo situation, the authors concluded that Pb species other than the free metal ion may have contributed to the Pb flux towards the cells, possibly involving the dissociation of labile Pb species, such as Pb-phosphate and Pb-labile complexes and the subsequent transport of the released free metal ions toward the intestinal membrane.

5.10.2.4 Alterations in Gastrointestinal Motility/Gastrointestinal Transit and Function

The effect of Pb on contractility of rat duodenum was determined in vivo in rats given an oral dose of Pb-acetate (44 mg/kg per day, Pb as 53 mM/L for 4 weeks) to investigate the possible mechanisms associated with Pb-induced abdominal colic and constipation (Karmakar and Anand, 1989). Deodenal motility and the amplitude of contractility of rat duodenum were decreased significantly in the Pb-exposed rats, leading the authors to conclude that there was a fundamental change in the contractility of the intestinal tract due to Pb intoxication.

Chronic Pb ingestion through drinking water (2-5 mg/mL, Pb-acetate for 55 days) caused a 20-fold increase in urinary excretion of D-ALA and an increase in blood Pb level (80 μg/dL), without any perturbations in propulsive motility of guinea pig colon (Rizzi et al., 1989). On the other hand, Lawrel et al. (1991) observed no changes in gastric contractions during ingestion in red-tailed hawks exposed to Pb-acetate (0.82 or 1.64 mg/kg body wt for 3 weeks). This low level of exposure has also been found to have no bearing on the regular passing of pellets of undigested material. Shraideh (1999) studied the effect of triethyl Pb-chloride on the rhythmic and peristalitic contractile activity of ileum isolated from Swiss mice. These authors observed no significant effect below 40 μM of TEL, while higher concentrations (40-120 μM) caused changes in contraction rhythm. These studies also reported that TEL above 120 μM induced irreversible changes in the ileal contractile activity. These and related studies are summarized in Table AX5-10.8.

5.10.2.5 Lead, Calcium, and Vitamin D Interactions in the Intestine

The complex biological interactions between Pb and calcium have been recognized and demonstrated in virtually every type of tissue. Studies of high-affinity Pb binding to intracellular

calcium receptors and transport proteins, as well as the involvement of Pb in calcium-activated and calcium-regulated processes, have added to our understanding of the effects of Pb on biological processes at the cellular level. The intestinal absorption of Pb is influenced by a variety of factors, including chemical and physical forms of the element, age at intake, and various nutritional factors. Work dating back to the 1940s established that the deposition of Pb in bone and soft tissue significantly increases under conditions of dietary calcium and phosphorus deprivation or by the administration of vitamin D to rachitic animals. Later, in the 1970s, it was demonstrated that dietary calcium status was a major contributing factor determining relative susceptibility to Pb intoxication.

Fullmer's group (Fullmer and Rosen, 1990; Fullmer, 1991, 1992, 1997) carried out a series of studies to investigate the potential interaction between calcium and Pb in the ingestion and intestinal absorption of Pb. Various parameters, such as absorption kinetics for Ca and Pb, activity of alkaline phosphatase, expression of the clabindin D gene, and the potential role of endocrine function in this interaction (as assessed by cholecalciferol and its active hormonal form, 1, 25-dihydroxycholecalciferol levels) were investigated. Fullmer and Rosen (1990) observed that chicks fed with low (0.5%) and adequate (1.2%) dietary calcium and exposed to Pb (0-0.8%) exhibited differential effects on intestinal Ca absorption depending on their dietary Ca status. In the chicks fed a low-calcium diet, Pb inhibited intestinal Ca absorption and calbindin D and alkaline phosphatase synthesis in a dose-dependent fashion. On the other hand, chicks fed the normal diet, showed no inhibition of Ca absorption. Based on these results, the authors postulated that Pb-induced alterations in intestinal Ca absorption may involve cholecalciferol and the endocrine system. In an extension of this study using young growing chicks, Fullmer (1991) observed similar results in 2-week Pb-exposed, but not in 1-week exposed, chicks.

As dietary Ca deficiency is associated with a marked increase in the body burden of Pb and in the susceptibility to Pb toxicity during chronic ingestion, Fullmer (1992) examined the effects of vitamin D supplementation on intestinal Pb and Ca absorption. When vitamin D-deficient chicks received physiologic amounts of vitamin D (0.1mg/day), intestinal ^{203}Pb and ^{47}Ca absorption rates were elevated by 4- and 8-fold, respectively. Along with this, calbindin D and alkaline phosphatase activities were also found to be significantly elevated. Ingestion of even the highest level of Pb (0.8 %) during the repletion phase had no effect on intestinal Ca absorption. To further understand the Pb-Ca interactions and the potential involvement of

vitamin D on intestinal absorption, Fullmer (1997) evaluated serum levels of 1, 25-dihydroxyvitamin D. Lead ingestion and Ca deficiency alone, or in combination, generally increased serum 1, 25-dihydroxyvitamin D levels over most of the ranges of Pb or Ca studied. However, in severe Ca deficiency, Pb ingestion resulted in marked decreases in serum 1, 25-dihydroxyvitamin D, intestinal Ca absorption, and calbindin D mRNA. From these studies using response surface models, Fullmer (1997) concluded that the interactions between Pb and Ca were mediated via changes in circulating 1, 25-dihydroxy vitamin D hormone, rather than via direct effects on the intestine.

Similar to Ca deficiency, iron deficiency has also been found to increase intestinal absorption of Pb, as indicated by increased blood and kidney Pb levels in iron-deficient rats exposed to dietary Pb; but the mechanistic details are not known (Crowe and Morgan, 1996). These and other related studies are summarized in Table AX5-10.9.

5.10.2.6 Lead and Intestinal Enzymes

Differential effects of Pb on intestinal brush border enzyme activity profiles were reported by Gupta et al. (1994). Across a concentration range of 0.5-6.0 mM, Pb-acetate was found to significantly inhibit Ca-Mg-ATpase, g-glutamyl transpeptidase, and acetylcholineesterase activities in a dose-dependent manner without effects on alkaline phosphatase.

Cremin et al. (2001) investigated the effects of oral succimer on the intestinal absorption of Pb in infant rhesus monkeys. These studies indicated that chelation therapy with DMSA for two successive 19-day periods significantly decreased GI absorption of Pb and increased urinary excretion of endogenous lead (see Table AX5-10.9).

5.10.2.7 Summary

- Gastrointestinal absorption of Pb is influenced by a variety of factors, including chemical and physical forms of the element, age at intake, and various nutritional factors. The degeneration of intestinal mucosal epithelium leading to potential malabsorption and alterations in the jejunal ultrastructure (possibly associated with distortion of glycocalyx layer) have been reported in the intestine of Pb-exposed rats.
- Lead in rat and bovine milk and, also, infant milk formula was demonstrated to be primarily associated with caseine micelles.
- Tracer studies using ^{203}Pb indicated that intragastric administration of Pb as a soluble salt resulted in Pb primarily accumulating in the deuodenum, regardless of dose or vehicle

used, whereas Pb from milk was found to be taken up by ileal tissue. Studies also suggested Pb ingestion through water was more toxic than ingestion through milk.

- Lead induced decreases in duodenal motility and amplitude of contractility of the intestinal tract has been reported for rats.
- Nutritional studies using various levels of Pb, Ca, and vitamin D in the diet indicate competition of Pb with Ca absorption. Supplementation with vitamin D has been reported to enhance intestinal absorption of Ca and lead. Physiological amounts of vitamin D administered to vitamin D-deficient rats resulted in elevated Pb and Ca levels. In the case of severe Ca deficiency, Pb ingestion results in a marked decrease in serum 1,25-hydroxy vitamin D.

Overall, our understanding of Pb effects on hepatic and gastro intestinal systems using in vitro cell culture models and in vivo animal models has increased greatly compared to the 1986 AQCD. Significant insights have emerged regarding the role of Pb in hepatic cholesterol synthesis, the role of inflammation in Pb-induced hepatotoxicity, and the contribution of newer chelation therapy in the amelioration of Pb-induced oxidative burden. Similarly, our knowledge has greatly enhanced as to the absorption, transport, and toxicity of Pb in the gastrointestinal tract.

5.11 LEAD-BINDING PROTEINS

Lead-binding proteins that are constitutively expressed within the cells and bind Pb can be classified into two types of protein. The first type of Pb-binding proteins are inducible, i.e., their concentration increases after exposure to Pb. The second type of Pb-binding proteins have binding sites that are saturable by Pb, but no discernible increase in protein content occurs after exposure to Pb. The second type is, perhaps, most pertinent to enzymes that can be inhibited by Pb.

The history of research on Pb-binding proteins dates back to 1936, when the presence of intranuclear inclusion bodies in the liver and kidney as manifestations of Pb poisoning was first described (Blackman, 1936). Later, detailed studies of the composition of renal tubular intranuclear Pb inclusion bodies and consequent alterations in mitochondrial structure and function followed.

5.11.1 Lead-Binding Proteins Within Intranuclear Inclusion Bodies in Kidney

Goyer (1968) examined the renal tubules of rats fed 1% Pb-acetate for up to 20 weeks, and found that dense, deeply staining intranuclear inclusions were located in the straight portion of the proximal tubules, accompanied by swollen, globular or ovoid, closely packed mitochondria with many marginated, irregular, or vesicular cristae. Accompanying these mitochondrial changes was the presence of generalized aminoaciduria. Goyer et al. (1968) also isolated mitochondria from Pb-exposed and control rats and demonstrated that mitochondria from the Pb-exposed rats showed reduced rates of respiration and oxidative phosphorylation.

Lead within the kidneys in Pb-poisoned rats was found to be concentrated in the nuclei and, within nuclei, in the nuclear inclusion body (Goyer et al., 1970a,b). Choie and Richter (1972) showed that rapid induction of inclusion bodies by injections of Pb salts in the rat resulted in cytoplasmic inclusions, suggesting that they were precursors to the intranuclear inclusions. Lead-containing nuclear inclusions were also found in organs other than the kidney, including liver and glial cells of the central nervous system (Goyer and Rhyne, 1973). Moore et al. (1973) dissolved the rat renal intranuclear inclusions in strong denaturing agents and found that the protein in the inclusions is acidic, with high levels of aspartic acid, glutamic acid, glycine, and cystine. Moore and Goyer (1974) later characterized the protein as a 27.5 kDa protein, which migrates as a single band on acrylamide gel electrophoresis. Repeated intraperitoneal injections of $CaNa_2EDTA$ resulted in the disappearance of the inclusion bodies in Pb-exposed rats, together with a marked decrease in kidney Pb levels (Goyer et al., 1978).

Shelton and co-workers have also explored the composition of Pb-binding proteins in the nuclear inclusion proteins of Pb-exposed rat kidneys. Shelton and Egle (1982) first described a 32 kDa protein with an isoelectric point of 6.3, which was isolated from the kidneys of rats treated with 1% Pb-acetate in rat chow or 0.75% Pb-acetate in drinking water for 13-17 weeks. In contrast to Goyer and co-workers, they used two-dimensional gel electrophoresis to isolate the protein from the nuclear inclusion bodies and demonstrated that it was present in Pb-exposed, but not control, kidneys (hence, inducible). This protein has been termed p32/6.3. Inhibitor studies with cycloheximide and actinomycin D (McLachlin et al., 1980; Choie et al., 1975) had indicated earlier that protein synthesis was required for induction of the nuclear and cytoplasmic inclusion bodies.

Egle and Shelton (1986) unexpectedly found that p32/6.3, now characterized by a monoclonal antibody, was constitutively present in the cerebral cortex, both in neurons and astrocytes. The protein was concentrated in the insoluble nuclear protein, findings similar as for the Pb-exposed kidney. Brain p32/6.3 was detected in rat, mouse, dog, man, and chicken. In rat brain, adult levels were achieved in 1 to 2 weeks after birth, whereas only trace amounts were found at 3 days. Brain p32/6.3 increased between postnatal days 10 to 12 in the guinea pig and days 15 to 21 in the rat, suggesting that the increase may be related in part to exposure to the external environment (Shelton et al., 1993). When neuroblastoma cells were cultured after 1- and 3-day exposure to Pb, the abundance of p32/6.3 increased. Simultaneous incubation with Pb and cycloheximide or actinomycin D increased in p32/6.3, suggesting that Pb selectively retards the degradation of the brain protein (Klann and Shelton, 1989). The amino acid composition of partially purified p32/6.3 revealed a high percentage of glycine, aspartic and glutamic acid (Shelton et al., 1990). Thus, the inducible protein, p32/6.3, can be extracted from nuclear inclusion bodies from the Pb-exposed rat kidney, and a similar or identical protein from adult rat brain. Whether the brain protein is constitutive or inducible by exposure to environmental Pb has yet to be determined.

Oskarsson and Fowler (1985) examined the influence of pretreatment with Pb by a single IP injection of Pb-acetate (50 mg Pb per kg) 1, 3, and 6 days before injecting ^{203}Pb. Rats were sacrificed 24 h later and the kidneys were examined both microscopically and for the distribution of ^{203}Pb. At 3 days, rat kidneys displayed fibrillar cytoplasmic inclusions, but at 6 days, these inclusions were less prominent and intranuclear inclusions were observed. ^{203}Pb uptake at 6 days was maximal in the purified nuclear fraction and in the nuclear inclusion bodies (7× and 20× control, respectively).

5.11.2 Cytoplasmic Lead-Binding Proteins in Kidney and Brain

The remaining studies of non-Pb-stimulated cytoplasmic kidney and brain Pb-binding proteins have been provided by Fowler and associates.

The first study (Oskarsson et al., 1982) reported on the Pb-binding proteins in kidney postmitochondrial cytosolic fractions. Binding of ^{203}Pb was found in two protein fractions of control kidneys with molecular weights of 11.5 and 63 kDa. Binding was markedly decreased after Pb pretreatment. The use of cadmium to stimulate metallothionein synthesis did not

increase ^{203}Pb binding to the 11.5 kDa protein. The two binding proteins were also present in brain, but not in liver or lung. Subsequently, Mistry et al. (1985) demonstrated three Pb-binding proteins (11.5, 63, and >200 kDa) in rat kidney cytosol, which had binding characteristics of high affinity, low capacity with respective K_d values of 13, 40, and 123 nM. The 11.5 kDa and, possibly, the 63 kDa proteins were capable of translocating Pb into the nucleus as shown by uptake of ^{203}Pb into nuclei incubated with tagged cytosolic proteins. Goering and Fowler (1984) showed that the 11.5 kDa protein, but not the 63 kDa protein was capable of reversing Pb-induced ALAD inhibition in liver homogenates. This effect was mediated both by chelation of Pb by the Pb-binding protein and by donation of zinc to ALAD (Goering and Fowler, 1985). Various divalent metal ions influence the binding of Pb to the rat kidney cytosolic binding proteins, with an order of displacement of Cd^{2+}>Zn^{2+}>Pb^{2+}. Ca^{2+} had no effect, while Fe^{2+} had a cooperative effect (Mistry et al., 1985). These observations may account for the previously demonstrated effect of concomitant Pb and cadmium administration in reducing total kidney Pb (Mahaffey et al., 1981) and preventing the development of intranuclear inclusion bodies (Mahaffey and Fowler, 1977).

Later studies by Fowler and Duval (1991) identified the rat renal Pb-binding protein as a cleavage product of α2-microglobulin, with a K_d of 10^{-8} M Pb. There are two forms of the protein in the kidney, differentiated by the cleavage of the first 9-N terminal residues from the higher-molecular weight form. Other studies by Smith et al. (1998) found two Pb-binding proteins in environmentally exposed human kidneys, identified as acyl-CoA binding protein (ACBP) or diazepam binding inhibitor (molecular weight 9 kDa) and thymosin β4 (molecular weight 5 kDa). These polypeptides have a high affinity for Pb (K_d~14 nM).

In rat brain, Goering et al. (1986) and Duval and Fowler (1989) explored the effects of environmental Pb on Pb-binding proteins and the ability of rat brain Pb-binding proteins to diminish the inhibition of hepatic ALAD by Pb (liver does not contain the Pb-binding protein). In the first study, a brain protein of 12 kDa was described, in comparison to the kidney Pb-binding protein of 9 kDa. Both competition of Pb binding between the brain Pb-binding protein and ALAD and donation of zinc by the brain protein (shown by ^{65}Zn uptake) were found to account for the decreased ALAD inhibition. In the second study the rat brain Pb-binding protein was described as having a molecular weight of 23 kDa, with significant levels of glutamic acid, aspartic acid, and cysteine. Polyclonal antibody to rat renal Pb-binding proteins

showed a lack of reactivity with the brain protein, indicating that the proteins are immunologically distinct.

Fowler et al. (1993) examined monkey kidney and brain from non-Pb-treated animals and isolated Pb-binding proteins that also had a relatively high content of aspartic and glutamic amino acid residues and were similar in size to the rat Pb-binding proteins. Polyclonal antibodies to α-2 microglobulin and metallothionein did not cross-react with either monkey kidney or brain proteins. Quintanilla-Vega et al. (1995) isolated a thymosin β4 and a second, as yet unidentified, protein with a molecular weight of 20 kDa and a pI of 5.9 from brains of environmentally Pb-exposed humans.

5.11.3 Lead-Binding Proteins in Erythrocytes

Intra-erythrocytic Pb-binding was initially attributed primarily to hemoglobin, molecular weight 64 kDa (Barltrop and Smith, 1972; Raghavan and Gonick, 1977; Ong and Lee, 1980; Lolin and O'Gorman, 1988), but more recent studies have ascribed the major Pb binding to ALAD, molecular weight 240–280 kDa. In contrast to this protein, several studies have focused on an inducible low molecular weight protein in workers chronically exposed to Pb and which seems to have a protective effect. The first recognition of this protein was by Raghavan and Gonick (1977) who found an approximately 10 kDa protein in Pb workers but not in controls, following Sephadex G-75 fractionation (Figure 5-11.1). Upon subsequent SDS-polyacrylamide gel electrophoresis, the protein split into two bands, only the uppermost of which contained Pb (Figure 5-11.2).

Raghavan et al. (1980) then went on to fractionate the erythrocyte Pb into a hemoglobin fraction, a 10 kDa fraction, free Pb, and a "residual Pb" fraction thought to be composed of membrane Pb and a high-molecular weight fraction. Lead workers manifesting toxicity at both high blood Pb and relatively low blood Pb levels showed high levels of residual Pb, attributed in the workers with toxicity at low blood leads to a very low quantity of the 10 kDa fraction. In a follow-up study, Raghavan et al. (1981) reported elevated levels of Pb in the high molecular weight fraction (pre-hemoglobin) and in the membrane fraction in workers with toxicity at both high and low BLLs. Again, those with toxicity at low blood Pb had low levels of the Pb bound to the 10 kDa protein. Membrane Pb was found to correlate inversely with membrane Na-K-ATPase; no correlation was seen with total blood Pb.

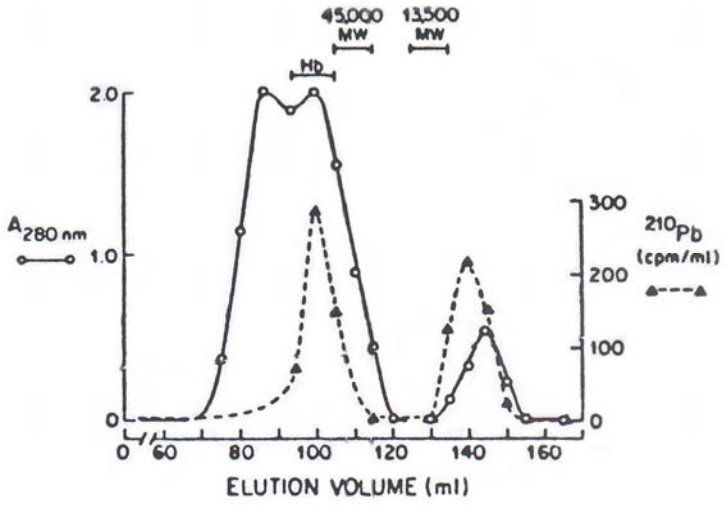

Figure 5-11.1. Sephadex G-75 gel filtration of RBC hemolysate from lead-exposed individual. Ultraviolet absorption and radioactivity of ^{210}Pb are plotted against elution volume. The column was calibrated with ovalbumin (mol wt 45,000) and ribonuclease (mol wt 13,700). Also indicated is the locus of hemoglobin (Hb). Hemolysates from normal control individuals showed no UV absorption or radioactivity in the volume eluting between 130 and 155 mL.

Source: Raghavan and Gonick (1977) with permission.

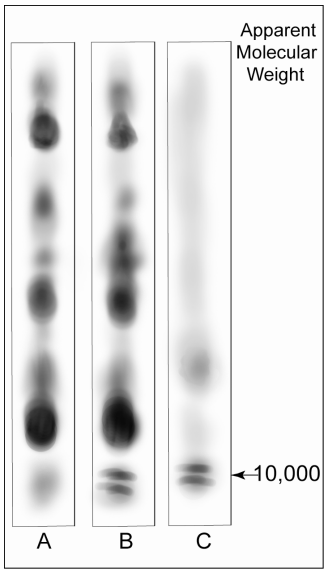

Figure 5-11.2. SDS-polyacrylamide gel electrophoresis of RBC hemolysates from normal control (A) and lead-exposed individuals (B), and of low-mol-wt. lead-binding protein (C). Stained with coomassie blue.

Source: Raghavan and Gonick (1977) with permission.

Gonick et al. (1985) partially purified the 10 kDa protein by HPLC using a protein I-125 column followed by isoelectric focusing on a sucrose gradient column. Three protein peaks resulted: one of 30 kDa, and two of 10 kDa. Only one of the latter peaks contained Pb. This peak had a pI of 5.3 and a molecular weight, determined by SDS-PAGE, of 12 kDa. The majority of Pb was found in this peak, which also contained calcium, zinc, and cadmium. Amino acid analysis showed a very high percentage of glycine (44%) and lower quantities of histidine, aspartic acid, and leucine.

Ong and Lee (1980) studied the distribution of ^{203}Pb in components of normal human blood. Ninety-four percent of ^{203}Pb was incorporated into the erythrocyte and 6% remained in the plasma. SDS-PAGE of plasma showed that 90% was present in the albumin fraction. Within the erythrocyte membrane, the most important binding site was the high molecular weight fraction, about 130–230 kDa. Within the erythrocytic cytoplasm, the protein band associated with ^{203}Pb had a molecular weight of 67 kDa as shown by the elution characteristics on G-75 chromatography. This was thought to be hemoglobin.

Lolin and O'Gorman (1988) and Church et al. (1993 a,b), following the same procedure as Raghavan and Gonick (1977), confirmed the findings of a low molecular weight protein in the erythrocytes of Pb workers, but not found in control patients. Lolin and O'Gorman (1988) quantitated the protein, which ranged from 8.2 to 52.2 mg/L RBC in Pb workers but found none in controls, again implying it to be an inducible protein. They found that the low molecular weight protein first appeared when the blood Pb concentration exceeded 39 µg/dL. A positive correlation was seen between the amount of the intra-erythocytic low molecular weight protein and dithiothreitol-activated ALAD activity but not the non-activated activity. Church et al. (1993a,b) also confirmed the findings of Raghavan et al. (1977). In 1993a, they described two patients with high blood Pb levels: an asymptomatic worker with a blood Pb of 180 µg/dL, and a symptomatic worker with a blood Pb of 161 µg/dL. In the first patient, approximately 67% of the erythrocyte Pb was bound to a low molecular weight protein of approximately 6–7 kDa. In the second patient, the protein only contained 22% of the total erythrocytic Pb. Church et al. (1993b) found that a sample of the low molecular weight protein purified from Pb workers, which they termed protein M, had characteristics of metallothionein, such as a molecular weight of 6.5 kDa, a pI between 4.7 and 4.9, and a greater UV absorbance at 254 nm than at 280 nm. Amino acid composition showed 33% cysteine but no aromatic amino acids. This composition

differed from that of the low molecular weight protein described by Gonick et al. (1985), which had a molecule weight of 12 kDa, a pI of 5.3, and amino acid analysis that showed no cysteine. This discrepancy might be explained by a combined Pb and cadmium exposure in the Church et al. (1993b) study, which may have produced a Pb-thionein.

Xie et al. (1998) used a Biogel A column instead of Sephadex G-75 to separate Pb-binding proteins from erythrocyte hemolysates from a control patient and from Pb-exposed workers. They clearly showed that the major Pb-binding was associated with a large molecular weight protein, consistent with ALAD, in both the controls and Pb workers. When they added increasing amounts of Pb to the blood of the control patient, a second low molecular weight protein peak occurred, in which Pb binding was larger than the ALAD peak (Figure 5-11.3). This second peak was also seen in a chronically Pb-exposed worker (Figure 5-11.4) and was estimated to be less than 30 kDa in molecular weight. Thus these results are consistent with the aforementioned studies.

5.11.4 Lead-Binding Proteins in Rat Liver

Sabbioni and Marafante (1976) explored the distribution of ^{203}Pb in rat whole tissue as well as in subcellular liver fractions. By far the largest quantity of Pb recovered was in the kidney, with lesser amounts in liver, spleen, and blood. Upon subcellular fractionation of the liver, the majority of ^{203}Pb was found in the nuclei, and most of the Pb was detected in the nuclear membrane fraction, bound exclusively to membrane proteins. The intranuclear Pb was associated with histone fractions. As reported by Oskarsson et al. (1982), Pb binding proteins were found in the cytoplasm of the liver.

5.11.5 Lead-Binding Proteins in Intestine

Fullmer et al. (1985) showed in the chick and cow that although Pb does not directly stimulate Pb-binding proteins in the intestine, Pb can displace calcium from calcium-binding proteins; and, thus, calcium-binding proteins may play a role in intestinal Pb transport. Purified calcium-binding protein from chick and cow, as well as calmodulin, troponin C, and oncomodulin were dialyzed against added labeled and unlabeled Pb or calcium. Results disclosed high affinity binding sites, with greater affinity for Pb than for calcium. Similar results

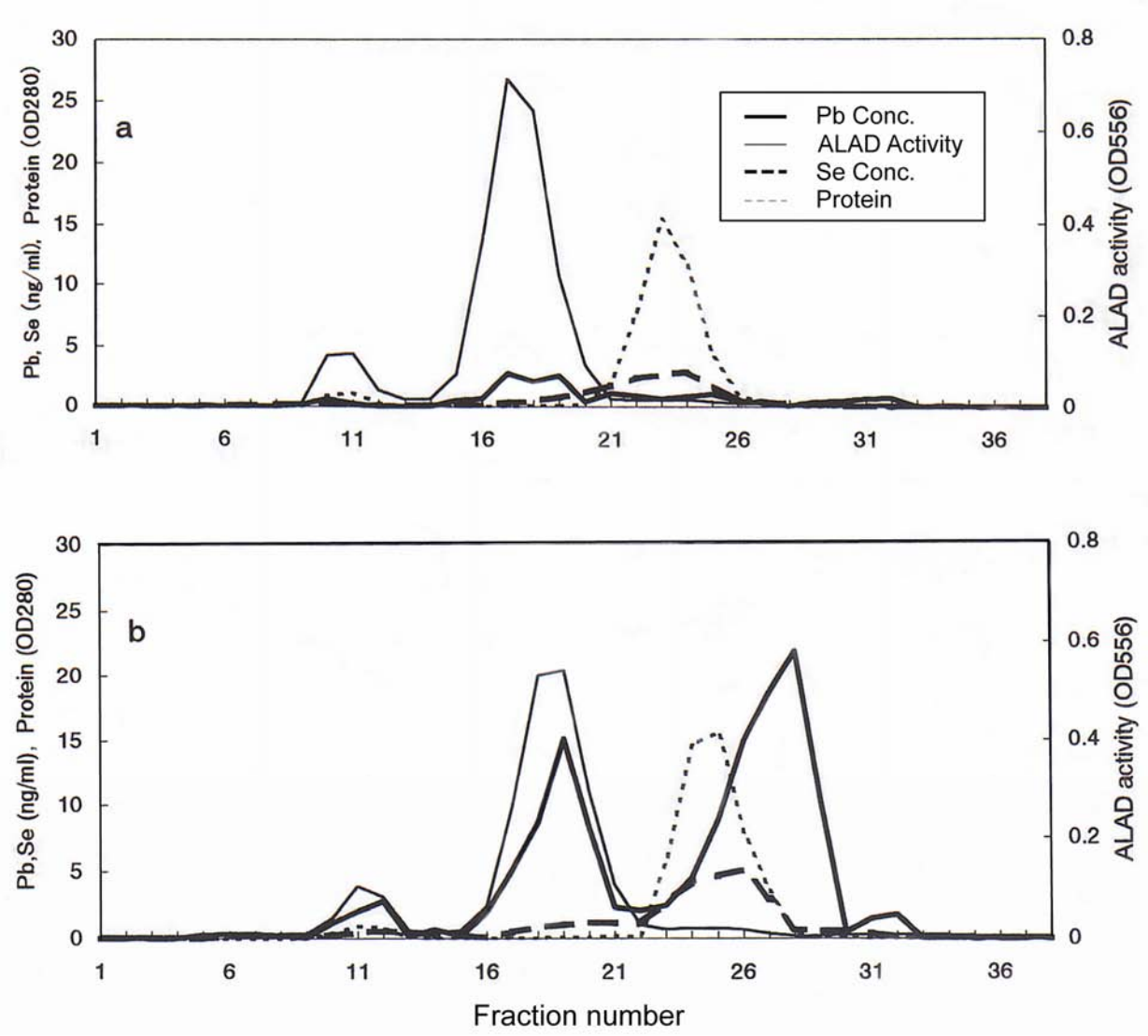

Figure 5-11.3. Chromatographic profiles of protein, ALAD activity and Pb in human erythrocytes incubated with 5% glucose solution containing Pb acetate. Blood was incubated (a) without Pb (b) 10 µM Pb (final concentrations).

Source: Adapted from Xie et al. (1998).

were obtained with calmodulin, troponin C, and oncomodulin, all members of the troponin C superfamily of calcium-binding proteins.

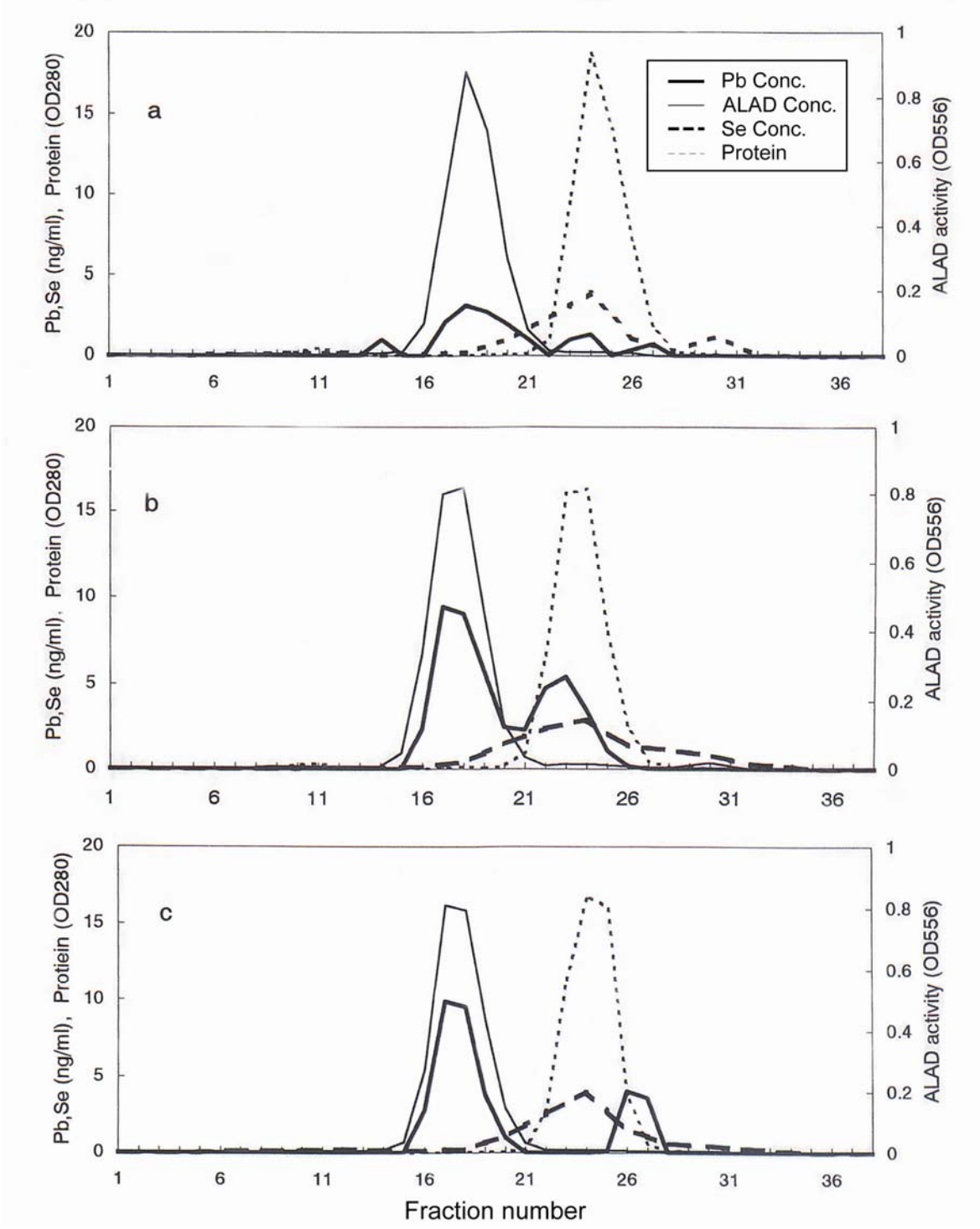

Figure 5-11.4. Chromatic profiles of protein, ALAD activity, Pb, and Se in the erythrocytes of lead-exposed workers. (a) control, (b) subacute exposure, (c) chronic exposure.

Source: Xie et al. (1998) with permission.

5.11.6 Relationship of Lead-Binding Protein to Metallothionein

Similarities of Pb-binding protein to metallothionein have been discussed earlier. Maitani et al. (1986) commented that hepatic zinc-metallothionein could be induced by intravenous and intraperitoneal injections of Pb into mice, but not by subcutaneous injection. Ikebuchi et al. (1986) found that a sublethal dose of Pb-acetate injected intraperitoneally into rats induced the synthesis of a Pb-metallothionein in addition to zinc-metallothionein. The Pb-metallothionein contained 28% half-cysteine and cross-reacted with an antibody against rat zinc-thionein II.

Goering and Fowler (1987 a,b) demonstrated that pretreatment of rats with zinc 48 and 24 h prior to injection of ^{203}Pb resulted in both zinc and Pb co-eluting with a zinc-thionein fraction on Sephadex G-75 filtration. In addition, both purified zinc-thionein-I and II bound ^{203}Pb in vitro. Gel filtration of incubates containing liver ALAD and ^{203}Pb demonstrated that the presence of zinc-thionein alters the cytosolic binding pattern of Pb, with less binding to ALAD. Zinc-thionein also donates zinc to activate ALAD. Goering and Fowler (1987b) found that pretreatment of rats with either cadmium or zinc affected liver ALAD activity when incubated with Pb. Liver and kidney zinc-thioneins, and to a lesser extent, cadmium, zinc-thionein decreased the free pool of Pb available to interact with ALAD, resulting in attenuated ALAD inhibition. Liu et al. (1991) further showed that zinc-induced metallothionein in primary hepatocyte cultures protects against Pb-induced cytotoxicity, as assessed by enzyme leakage and loss of intracellular potassium.

Qu et al. (2002) and Waalkes et al. (2004) have shown that metallothionein-null phenotypic mice are more susceptible to Pb injury over a 20-week period than wild type mice. Unlike the wild type mice, Pb-treated metallothionein-null mice showed nephromegaly and significantly decreased renal function after exposure to Pb. The metallothionein-null mice accumulated less renal Pb than wild type and formed no inclusion bodies. When the observations were extended to 104 weeks, renal proliferative lesions (adenoma and cystic tubular atypical hyperplasia) were more common and severe in metallothionein-null than in wild type mice. A metastatic renal cell carcinoma occurred in a metallothionein-null mouse, whereas none occurred in wild type mice. Such studies lend credence to the view that metallothinein, or a closely related gene, is involved in the formation of Pb-binding proteins in the kidney.

5.11.7 Is ALAD an Inducible Enzyme and is it the Principal Lead-Binding Protein in the Erythrocyte?

The enzyme ALAD has been found to be the most sensitive indicator of Pb exposure and toxicity (Granick et al., 1973, Buchet et al., 1976). In the 1980s, two articles were presented appearing to show that ALAD is inducible after Pb exposure in humans. By comparing a nonexposed control population of Pb workers and assaying ALAD by means of immunoassay or as 'restored' ALAD activity (i.e., incubation with heat, zinc and dithiothreitol) both articles indicated that the amount of ALAD, as contrasted to ALAD activity, was increased by Pb exposure (Fujita et al., 1982; Boudene et al., 1984). Similar findings were reported for the rat (Fujita et al., 1981). Subsequent studies have focused on the effect of ALAD polymorphism on the susceptibility to Pb intoxication. ALAD is a zinc-containing enzyme, which catalyzes the second step of heme synthesis, i.e., catalyzes the condensation of two delta-aminolevulinic acid molecules into one molecule of porphobilinogen (Boudene et al., 1984). It is a polymorphic protein with three isoforms: ALAD-1, ALAD 1-2, and ALAD 2-2. Several studies have shown that, with the same exposure to Pb, individuals with the ALAD-2 gene have higher blood Pb levels (Astrin et al., 1987; Wetmur, 1994; Wetmur et al., 1991; Smith et al., 1995a; Bergdahl et al., 1997; Perez-Bravo et al., 2004; Kim et al., 2004). Initially it was thought that these individuals might be more susceptible to Pb poisoning (Wetmur et al., 1991), but it is now appreciated that the ALAD-2 gene offers protection against Pb poisoning by binding Pb more securely (Kelada et al., 2001). In support of this statement, it can be cited that individuals with the ALAD 1-2/2-2 genotypes, in comparison to those with the ALAD 1-1 genotype, have not only higher blood Pb but also decreased plasma levulinic acid (Schwartz et al., 1997), lower zinc protopophyrin (Kim et al., 2004), lower cortical bone Pb (Smith et al., 1995b), and lower amounts of DMSA-chelatable Pb (Schwartz et al., 1997, 2000).

The significance of erythrocyte ALAD binding to Pb was initially confirmed by a study by Bergdahl et al. (1997) in which the authors used a FPLC Superdex 200 HR 10/30 chromatographic column coupled to ICP-MS (for determination of Pb) to examine erythrocytes from Pb workers and controls. They found the principal Pb-binding protein peak to be 240 kDa (rather than the presumed hemoglobin peak reported by Barltrop and Smith (1972) and Raghavan and Gonick (1977), using Sephadex G-75 chromatography). This was shown to be ALAD by binding to specific ALAD antibodies. Two additional smaller Pb-binding peaks of

45 kDA and 10 kDa were also seen, but not identified. Bergdahl et al. (1997) attributed the discrepancies in the studies to the fact that Sephadex G-75 separates proteins in the range of 3 to 80 kDa, making the separation of hemoglobin (molecular weight 64 kDa) from ALAD (molecular weight 240–280 kDa) very difficult. In addition, the earlier studies had utilized binding of ^{203}Pb or ^{210}Pb to identify the binding proteins, a technique which may have skewed the findings if ALAD were already saturated. ALAD binding capacity for Pb has been measured at 85 µg/dL in erythrocytes or 40 µg/dL in whole blood (Bergdahl et al., 1998), which would permit a greater degree of binding to the low molecular weight component when blood Pb exceeded 40 µg/dL. Bergdahl et al. (1998) have speculated that the low molecular weight component might be acyl-CoA-binding protein, identical to the kidney Pb-binding protein described by Smith et al. (1995b). Goering and Fowler (1987) had reported earlier that the presence of low molecular weight high affinity (K_d 10^{-8}M) Pb-binding proteins in kidney and brain served as protection against ALAD inhibition in those organs, whereas the absence of the low molecular weight proteins in liver contributed to the greater sensitivity to ALAD inhibition in that organ.

A summary of the findings on Pb-binding protein can be found in Table AX5-11.1.

5.11.8 Summary

- There appears to be a consensus that the enzyme, ALAD, a 280 kDa protein, is inducible and is the major Pb-binding protein within the erythrocyte. ALAD polymorphism influences the degree of Pb-binding as the ALAD-2 phenotype binds more Pb in a nontoxic fashion than ALAD-1. What is more confusing is the nature and importance of the low molecular weight erythrocytic Pb-binding protein. There is no doubt that it appears in Pb-exposed workers but not in controls and that its molecular weight is approximately 10 kDa. The in vitro addition of Pb to erythrocytes of controls results in progressively increasing Pb binding to a low molecular weight protein peak migrating in the same position as the low molecular weight protein from Pb workers. This confirms the fact that once the binding capacity of ALAD is saturated, Pb shifts to the low molecular weight protein. The nature of the low-molecular weight protein is also questionable, it has been variously identified as a 12 kDa protein with a high percentage of glycine plus histidine, aspartic acid, and leucine and as a 6.5 kDa molecule with a large percentage of cysteine and a greater UV absorbance at 254 than 280 nm. The latter findings suggest that the protein might be a metallothionein.

- Metallothionein is a protein that is mildly inducible by Pb but to a much greater degree by zinc and cadmium. What is more significant is that Pb binds to pre-formed metallothionein, stimulated by zinc or cadmium, so that under these conditions a

Pb-thionein forms. Thus, concomitant Pb and cadmium exposure occurred in Pb workers that could account for the finding of a metallothionein-like protein in those workers.

- Extensive studies of cytoplasmic Pb-binding proteins in non-Pb-treated rats, human, and monkeys have been reported. The Pb-binding protein in rat kidney has been identified as a cleavage product of α-2 microglobulin. The low molecular weight Pb-binding proteins in human kidney have been identified as thymosin β4 (molecular weight 5 kDa) and acyl-CoA binding protein (molecular weight 9 kDa). In human brain the Pb-binding proteins were thymosin β4 and an unidentified protein of 23 kDa. Antibodies to α-2 microglobulin and metallothionein did not cross-react with monkey kidney or brain Pb-binding proteins, suggesting species differences. Whether the low molecular weight human kidney and brain Pb-binding proteins are similar or identical to the low molecular weight Pb-binding proteins in erythrocytes is at present unknown. Perhaps some clarification would be provided were subsequent investigators to contrast normal with Pb-exposed rats and to measure the resting and inducible Pb-binding protein levels in kidney, brain, and erythrocyte.
- The possible role of metallothionein as a renal Pb-binding protein assumes greater importance because of the work showing that metallothionein-null mice failed to respond to Pb exposure by developing intranuclear Pb inclusion bodies or greatly increased Pb content of the kidneys.

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6. EPIDEMIOLOGIC STUDIES OF HUMAN HEALTH EFFECTS ASSOCIATED WITH LEAD EXPOSURE

6.1 INTRODUCTION

This chapter assesses information regarding the biological effects of lead exposure, with emphasis on (1) qualitative characterization of lead-induced effects and (2) delineation of concentration-response relationships for key health effects of most concern. Epidemiologic studies linking lead exposure to health effects were assessed in the 1986 Air Quality Criteria for Lead (U.S. Environmental Protection Agency, 1986a), an associated addendum (U.S. Environmental Protection Agency, 1986b), and a 1990 Supplement (U.S. Environmental Protection Agency, 1990). Many earlier studies reported lead effects on child development (psychometric intelligence), blood pressure and related cardiovascular endpoints, heme biosynthesis, kidney, and reproduction and development. Numerous more recent epidemiologic studies discussed in this chapter have further evaluated these relationships to lead exposure, thereby providing an expanded basis for assessment of health effects associated with exposure to lead at concentrations currently encountered by the general U.S. population.

Special emphasis is placed here on discussion of the effects of lead exposure in children. Children are particularly at risk due to sources of exposure, mode of entry, rate of absorption and retention, and partitioning of lead in soft and hard tissues. The greater sensitivity of children to lead toxicity, their inability to recognize symptoms, and their dependence on parents and healthcare professionals make them an especially vulnerable population requiring special consideration in developing criteria and standards for lead.

As discussed elsewhere in this document (Chapter 5), extensive experimental evidence links lead exposure with health effects in laboratory animals. Thus, many of the reported epidemiologic associations of lead health effects have considerable biological credibility. Accordingly, the new epidemiologic studies of lead assessed here are best considered in combination with information from the other chapters on lead exposure and on toxicological effects of lead in animals. The epidemiologic studies constitute important information on associations between health effects and exposures of human populations to “real world” lead concentrations and also help to identify susceptible subgroups and associated risk factors.

6.1.1 Approach to Identifying Lead Epidemiologic Studies

Numerous lead epidemiologic papers have been published since completion of the 1986 Lead AQCD/Addendum, and 1990 Supplement. A systematic approach has been employed to identify relevant new epidemiologic studies for consideration in this chapter. In general, an ongoing literature search has been used in conjunction with other strategies to identify lead epidemiologic literature pertinent to developing criteria for the National Ambient Air Quality Standards (NAAQS) for lead. A publication base was established using Medline, Pascal, BIOSIS, and Embase, and a set of search terms aimed at identifying pertinent literature.

While the above search regime accessed much of the pertinent literature, additional approaches augmented such traditional search methods. For example, a Federal Register Notice was issued requesting information and published papers from the public at large. Also, non-EPA chapter authors, expert in this field, identified literature on their own, and EPA staff also identified publications as part of their assessment and interpretation of the literature. Lastly, additional potentially relevant publications are expected to be identified and included as a result of external review of this draft document by the public and CASAC. The principal criteria used for selecting literature for the present assessment is to focus mainly on those identified studies that evaluate relationships between health outcome and lead exposure at concentrations in the range of those currently encountered in the United States. New studies published or accepted for publication through June 2005, as identified using the approaches above, have been included in this draft lead air quality criteria document (Lead AQCD), and additional efforts are being made to identify and assess more recent studies.

6.1.2 Approach to Assessing Epidemiologic Evidence

Epidemiologic studies have evaluated lead effects on a wide range of health endpoints that include, but are not limited to: neurotoxic effects (e.g., psychometric intelligence, behavioral disturbances, and neurodevelopmental deficits), renal effects, cardiovascular effects, reproductive and developmental effects, genotoxic and carcinogenic effects, and immune effects. The epidemiologic strategies most commonly used in lead health studies are: (1) cross-sectional studies that examine the exposure and health outcome at a single point in time; and (2) prospective longitudinal cohort studies that follow a group of individuals over time. Both of these are types of observational rather than experimental studies.

An overall approach useful for assessing epidemiologic evidence was stated in the 2004 PM AQCD (U.S. Environmental Protection Agency, 2004), as summarized here. That is, the critical assessment of epidemiologic evidence presented in this chapter is conceptually based upon consideration of salient aspects of the evidence of associations so as to reach fundamental judgments as to the likely causal significance of the observed associations (see Hill, 1965). The general evaluation of the strength of the epidemiologic evidence reflects consideration not only of the magnitude and precision of reported lead effect estimates and their statistical significance, but also of the robustness of the effects associations. Statistical significance corresponds to the allowable rate of error (Type I error) in the decision framework constructed from assuming that a simple null hypothesis of no association is true. It is a conditional probability; for statistical significance, typically there is a less than 0.05 chance of rejecting the null hypothesis given that it is true. Robustness of the associations is defined as stability in the effect estimates after considering a number of factors, including alternative models and model specifications, potential confounding by copollutants, as well as issues related to the consequences of measurement error.

Consideration of the consistency of the effects associations, as discussed in the following sections, involves looking across the results obtained by various investigators in different locations and times. Relevant factors are known to exhibit much variation across studies, e.g., (1) presence and levels of other toxicants or pollutants of concern and (2) relevant demographic factors related to sensitive subpopulations. Thus, consideration of consistency is appropriately understood as an evaluation of the similarity or general concordance of results, rather than an expectation of finding quantitative results within a very narrow range.

Looking beyond the epidemiologic evidence, evaluation of the biological plausibility of the lead-health effects associations observed in epidemiologic studies reflects consideration of both exposure-related factors and dosimetric/toxicologic evidence relevant to identification of potential biological mechanisms underlying the various health outcomes. These broader aspects of the assessment are only touched upon in this chapter but will be more fully integrated in discussions presented in Chapter 7 (Integrative Synthesis).

In assessing the relative scientific quality of epidemiologic studies reviewed here and to assist in interpreting their findings, the following considerations were taken into account:

(1) To what extent are the biological markers used of adequate quality and sufficiently representative to serve as credible exposure indicators, well-reflecting geographic or temporal differences in study population exposures?

(2) Were the study populations well defined and adequately selected so as to allow for meaningful comparisons between study groups or meaningful temporal analyses of health effects results?

(3) Were the health endpoint measurements meaningful and reliable, including clear definition of diagnostic criteria utilized and consistency in obtaining dependent variable measurements?

(4) Were the statistical analyses used appropriate, as well as being properly performed and interpreted?

(5) Were likely important covariates (e.g., potential confounders or effect modifiers) adequately controlled for or taken into account in the study design and statistical analyses?

(6) Were the reported findings internally consistent, biologically plausible, and coherent in terms of consistency with other known facts?

These guidelines provide benchmarks for judging the relative quality of various studies and in assessing the body of epidemiologic evidence. Detailed critical analysis of all epidemiologic studies on lead health effects, especially in relation to all of the above questions, is beyond the scope of this document.

6.1.3 Considerations in the Interpretation of Epidemiologic Studies of Lead Health Effects

Prior to assessing results from recent lead epidemiologic studies, issues and questions arising from study designs and analysis methods used in the evaluation of lead health effects are briefly discussed. Study design can restrict the health effect parameters that can be estimated. Separate considerations need to be made for acute versus chronic effect studies, as well as individual versus aggregate-level analyses. Issues include measurement error, the functional form of relationships (especially at low exposure levels) and the potential for confounding. Aspects of these issues are briefly noted below, then are considered as various studies are reviewed in the following sections on specific health effect endpoints. Finally, they are further examined as part of the interpretive assessment(Section 6.9) at the end of this chapter.

Measurement error is an important factor to consider, both for measurement of the health effect outcome and the representativeness of the biomarkers of exposure (principally blood and bone lead) used in most key epidemiologic studies. For health outcome measures, the reliability and validity of the measurement need to be assessed. In addition, the appropriateness of the outcome measure for studying the hypothesis of interest needs to be determined. The critical issues of outcome measurement and classification are, to some extent, endpoint-specific, and will thusly be discussed further in the individual sections.

Exposure misclassification can result in a notable reduction of statistical power in studies, especially in those that focus on the lower end of the exposure range. Limitations of blood lead as an exposure index include the use of a single blood lead concentration to represent lead body burden. Also of concern is the most relevant blood sample collection time point for to use in evaluating possible associations with health outcomes (e.g., at 2 years of age when peak lead exposure is expected versus concurrent blood lead samples). Another consideration is that similar blood lead concentrations in two individuals do not necessarily reflect similar body burdens. An added complication is that the relationship between lead intake and blood lead concentration appears to be curvilinear. Bone lead determinations are typically considered a measure of longer-term lead exposure; but, the X-ray fluorescence (XRF) method typically used to assess lead levels in bone also has limitations, including the relatively high minimum detection limit. The type of bone measured to determine lead exposure is another important aspect.

The relationship between a measurement of a health outcome endpoint and an estimate of lead exposure based on a biomarker is an important concept. Modeling this relationship provides a numerical slope that quantifies the relationship between lead exposure and health outcome. These models must address differences in the relationship at different concentration ranges of exposure and present the functional form that best describes such data. Various models, both linear and nonlinear, have been considered to examine lead exposure-health effect relationships. This is especially important at low lead exposures. For example, a curvilinear relationship has been reported for neurodevelopmental and cardiovascular outcomes at low lead exposure levels.

Depending on the subjects being examined for lead exposure effects, various other factors can lead to confounding of the relationship being considered. Potential confounding factors largely depend on the health outcome of interest and the study population. Some potential

confounding factors in children, for whom the major health concerns include neurological and developmental deficiencies, include: socioeconomic status (SES); nutritional status; quality of home environment (e.g., HOME score); parental education; parental IQ; and birth weight, as a few examples. For adults, factors that may confound the association between lead and cardiovascular health outcomes include: age; diet; alcohol use; smoking; and potential for copollutant exposures, such as cadmium. Control for potential confounding factors can be attempted at the study design phase and/or during statistical analysis.

6.1.4 Approach to Presenting Lead Epidemiologic Evidence

In the main body of this chapter, each section starts by concisely highlighting important points derived from the 1986 Lead AQCD/Addendum, and the 1990 Supplement. Particular emphasis is focused on studies and analyses that provide pertinent information for the critical assessment of health risks from lead exposure. Not all studies are accorded equal weight in the overall interpretive assessment of evidence regarding lead-associated health effects. Among well-conducted studies with adequate control for confounding, increasing scientific weight is accorded in proportion to the precision of their effect estimates. To ensure a thorough appraisal of the evidence, more detailed information on key features (including study design, analysis, lead biomarkers of exposure, and health outcome results) of important new studies are summarized in tables in the Annex for this Chapter 6 (Annex AX6).

Emphasis is placed on main body text discussion below of (1) new studies employing standardized methodological analyses for evaluating lead effects across several cities and providing overall effect estimates based on combined analyses of information pooled across multiple cities; (2) studies conducted in the U.S. or Canada; and (3) meta-analyses of individual studies conducted in various cities. Multicity studies are of particular interest and value due to their evaluation of a wider range of lead exposures and large numbers of observations, thus generally providing more precise effect estimates than most smaller scale studies of single cities. Furthermore, multicity studies have the potential to provide especially valuable evidence regarding relative homogeneity and/or heterogeneity of lead health effects relationships across geographic locations. The potential impacts of the underlying health status of populations and cultural differences in the case of intelligence testing (one of the major health outcomes in children) also need to be accounted for in the assessment; thus, U.S. studies are emphasized over

non-U.S. studies. In accordance with the emphasis placed on the lead epidemiologic studies in this chapter, Chapter 6 Annex tables are organized by region, with multicity studies in each region presented first.

In the ensuing sections, epidemiological studies of biological markers of lead exposure are discussed first, in Section 6.2. The neurotoxic effects of lead are next discussed in Section 6.3 for children and adults, followed by discussion of the renal and cardiovascular effects of lead in Sections 6.4 and 6.5. Section 6.6 then discusses reproductive and developmental effects of lead, and Section 6.7 discusses genotoxic and carcinogenic effects of lead. Section 6.8 discusses the effects of lead on the immune system. The effects of lead on other organ systems (including the hematopoietic, endocrine, hepatic, gastrointestinal, and respiratory systems) are assessed in Section 6.9. Effects of lead on bone and teeth, as well as on ocular health are also discussed in Section 6.9. Finally, Section 6.10 provides an interpretative assessment of the overall epidemiologic evidence for lead health effects.

6.2 BIOLOGICAL MARKERS OF LEAD BODY BURDEN AND EXPOSURE

6.2.1 Lead in Blood

6.2.1.1 Summary of Key Findings from the 1986 Lead AQCD

The extensive use of blood lead concentration as a dose metric reflects mainly the greater feasibility of incorporating blood lead measurements into clinical or epidemiologic studies, compared to other potential dose indicators, such as lead in kidney, plasma, urine, or bone (Flegal and Smith, 1995; Graziano, 1994; Skerfving, 1988). However, blood lead measurements have several limitations as measures of lead body burden and exposure that relate to the kinetics of blood lead in relation to exposure and body burden. These limitations were noted in Section 13.3.2 of the 1986 Lead AQCD, which discusses attributes and limitations of blood lead concentration as an indicator of internal exposure. Relevant developments since the 1986 Lead AQCD was completed include numerous studies of determinants of lead levels in bone (see Section 6.2.2), which provide further support for the importance of this relatively slow kinetic compartment in assessing the blood lead concentration as an index of lead exposure. The enhanced understanding of lead biokinetics has also been consolidated into exposure-biokinetics

models (see Chapter 4), which not only serve to illustrate exposure-blood-body burden relationships, but also provide a means for making predictions about these relationships that can be tested experimentally or in epidemiologic studies. The basic concepts laid out in the 1986 Lead AQCD, that the concentration of lead in blood is largely determined by the relatively recent exposure history of the individual and that it reflects the level of lead in a relatively mobile and small compartment, remain valid. Especially in children, who experience a more rapid turnover of bone mineral, an endogenous source of lead, blood lead concentrations closely parallel changes in total body burden.

6.2.1.2 Analytical Methods for Measuring Lead in Blood

Analytical methods for measuring lead in blood include flame atomic absorption spectrometry (AAS), graphite furnace atomic absorption spectrometry (GFAAS), anode stripping voltammetry (ASV), inductively coupled plasma-atomic emission spectroscopy (ICP-AES), and inductively coupled plasma-mass spectrometry (ICP-MS). GFAAS and ASV are generally considered to be the methods of choice (Flegal and Smith, 1995). Background correction, such as Zeeman background correction that minimizes the impact of the absorbance of molecular species, must be applied. Limits of detection for lead using AAS are on the order of 5-10 μg/dL for flame AAS measurements, approximately 0.1 μg/dL for flameless AAS measurements, and 0.005 μg/dL for GFAAS (Flegal and Smith, 1995; National Institute for Occupational Safety and Health, 1994). A summary of standard methods that have been reported for blood lead analysis are provided in Annex Table AX6-2.1. Sample preparation usually consists of wet ashing in heated strong acid (National Institute for Occupational Safety and Health, 1977a,b,c,d,e); however, preparation methods not requiring wet ashing have also been reported (Aguilera de Benzo et al., 1989; Delves and Campbell, 1988; Manton and Cook, 1984; National Institute for Occupational Safety and Health, 1977f; Que Hee et al., 1985; Zhang et al., 1997). The presence of phosphate, ethylenediaminetetraacetic acid (EDTA), or oxalate can sequester lead and cause low readings in flame AAS (National Institute for Occupational Safety and Health, 1984). A comparison of IDMS, ASV, and GFAAS showed that all three of these methods can be used to quantify lead levels in blood (Que Hee et al., 1985).

6.2.1.3 Levels of Lead in Blood

Blood lead concentrations in the U.S. general population have been monitored in the National Health and Nutrition Examination Survey (NHANES) conducted by the Centers for Disease Control and Prevention. Data from the most recent survey (NHANES IV, Centers for Disease Control, 2005) are shown in Tables 6-2.1 and 6-2.2. For survey years 2001-2002, the geometric mean blood lead concentration for ages >1 year (n = 8,945) was 1.45 μg/dL (95% CI: 1.39, 1.52); with the geometric mean in males (n = 4,339) being 1.78 μg/dL (95% CI: 1.71, 1.86) and in females (n = 4,606) being 1.19 μg/dL (95% CI: 1.14, 1.25). Blood lead concentrations in the U.S. general population have decreased over the past three decades as regulations regarding lead paint, leaded fuels, and lead-containing plumbing materials have decreased exposure. Changes over time in children are shown in Figure 6-2.1.

Yassin et al. (2004) analyzed occupational category strata from NHANES III (1988–1994; Table 6-2.3). The geometric mean for all adults (n = 11,126) included in the analysis was 2.42 μg/dL (GSD 6.93), with the highest means estimated for vehicle mechanics (n = 169; GM 4.80 μg/dL [GSD 3.88]) and construction workers (n = 122; GM 4.44 μg/dL [GSD 7.84]). See Annex Table AX6-2.2 for a summary of selected measurements of blood lead levels in humans.

6.2.1.4 Blood Lead as a Biomarker of Lead Body Burden

A simple conceptual representation of the lead body burden is that it is comprised of a fast turnover pool, comprised mainly of soft tissue, and a slow pool, comprised mainly of skeletal tissues (Rabinowitz et al., 1976; see Chapter 4 for detailed discussion of this and other more complex models of lead biokinetics). The rapid pool has an elimination half-life of ~28 days and comprises <1% of the lead body burden. The slow pool has an elimination half-life of several decades and comprises >90% of the total lead body burden. Blood, which comprises ~1% of body burden, exchanges with both the slow and fast pools, and exhibits multiphasic elimination kinetics. The dominant phase, exhibited shortly after a change in exposure occurs, has a half-life of ~20–30 days. A slower phase becomes evident with longer observation periods following a

Table 6-2.1. Blood Lead Concentrations in U.S. by Age, NHANES IV (1999–2002)

Age	1–5 years		6–11 years		12–19 years		≥20 years	
Survey Period	*1999–2000*	*2001–2002*	*1999–2000*	*2001–2002*	*1999–2000*	*2001–2002*	*1999–2000*	*2001–2002*
N	723	898	909	1,044	2,135	2,231	4,207	4,772
Blood Lead (µg/dL)[a]	2.23 (1.96, 2.53)	1.70 (1.55, 1.87)	1.51 (1.36, 1.66)	1.25 (1.14, 1.36)	1.10 (1.04, 1.17)	0.94 (0.90, 0.99)	1.75 (1.68, 1.81)	1.56 (1.49, 1.62)

[a]Blood lead concentrations presented are geometric means (95% CI).

Table 6-2.2. Blood Lead Concentrations in U.S. by Gender, NHANES IV (1999–2002)

Gender	Males		Females	
Survey Period	1999–2000	2001–2002	1999–2000	2001–2002
n	3,913	4,339	4,057	4,606
Blood Lead (µg/dL)[a]	2.01 (1.93, 2.09)	1.78 (1.71, 1.86)	1.37 (1.32, 1.43)	1.19 (1.14, 1.25)

[a]Blood lead concentrations presented are geometric means (95% CI).

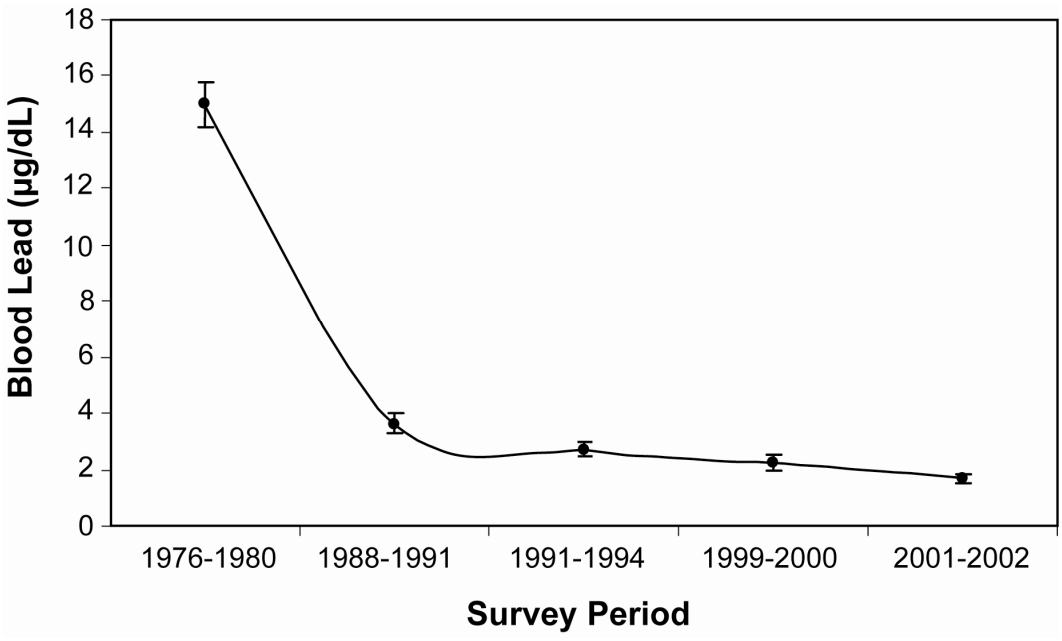

Figure 6-2.1. Blood lead concentrations in U.S. children, 1-5 years of age. Shown are geometric means and 95% confidence intervals as reported from the NHANES II (1976–1980) and NHANES III Phase 1 (1988–1991; Pirkle et al., 1994); NHANES III Phase 2 (1991–1994; Pirkle et al., 1998); and NHANES IV (1999-2000, 2001-2002; Centers for Disease Control, 2005).

decrease in exposure. The half-life of this slow phase has been estimated to be ~3 to 30 years and appears to correlate with finger bone lead levels. This characterization is supported by measurements of lead contents of cadaver tissues (Barry, 1975; Schroeder and Tipton, 1968), lead isotope kinetics in adults (Chamberlain et al., 1978; Rabinowitz et al., 1976; Griffin et al., 1975), and measurements of blood and bone lead levels in retired lead workers (Schütz et al., 1987; Christoffersson et al., 1986).

As a consequence of a relatively large fraction of the body burden having a relatively slow turnover compared to blood, a constant lead uptake (or constant intake and fractional absorption) gives rise to a quasi-steady state blood lead concentration, while the body burden continues to increase, largely as a consequence of retention of lead in bone (Figure 6-2.2). As a result, the contribution of blood lead to body burden decreases over time. An abrupt change in lead uptake gives rise to a relatively rapid change in blood lead, to a new quasi-steady state, achieved in ~75-100 days (i.e., 3-4 times the blood elimination half-life). In the hypothetical simulation shown in Figure 6-2.2, body burden has approximately doubled (from 5 to 10 mg) as a result of a

Table 6-2.3. Blood Lead Concentrations by Occupation, NHANES III (1988-1994)

Occupation	n	Blood Lead (µg/dL)		
		GM	GSD	Maximum
Vehicle mechanics	169	4.80	3.88	28.1
Food service workers	700	2.00	2.69	27.0
Management, professional, technical, and sales workers	4,768	2.13	4.05	39.4
Personal service workers	1,130	2.48	4.52	25.9
Agricultural workers	498	2.76	4.02	23.4
Production workers: machine operators, material movers, etc.	1,876	2.88	4.24	52.9
Laborers other than in construction	137	3.47	3.36	21.8
Transportation workers	530	3.49	5.19	22.3
Mechanics other than vehicle mechanics	227	3.50	4.91	16.6
Construction trades people	470	3.66	4.64	16.9
Construction laborers	122	4.44	7.84	36.0
Health service workers	499	1.76	2.24	22.4
All workers	**11,126**	**2.42**	**6.93**	**52.9**

Data from Yassin et al. (2004).

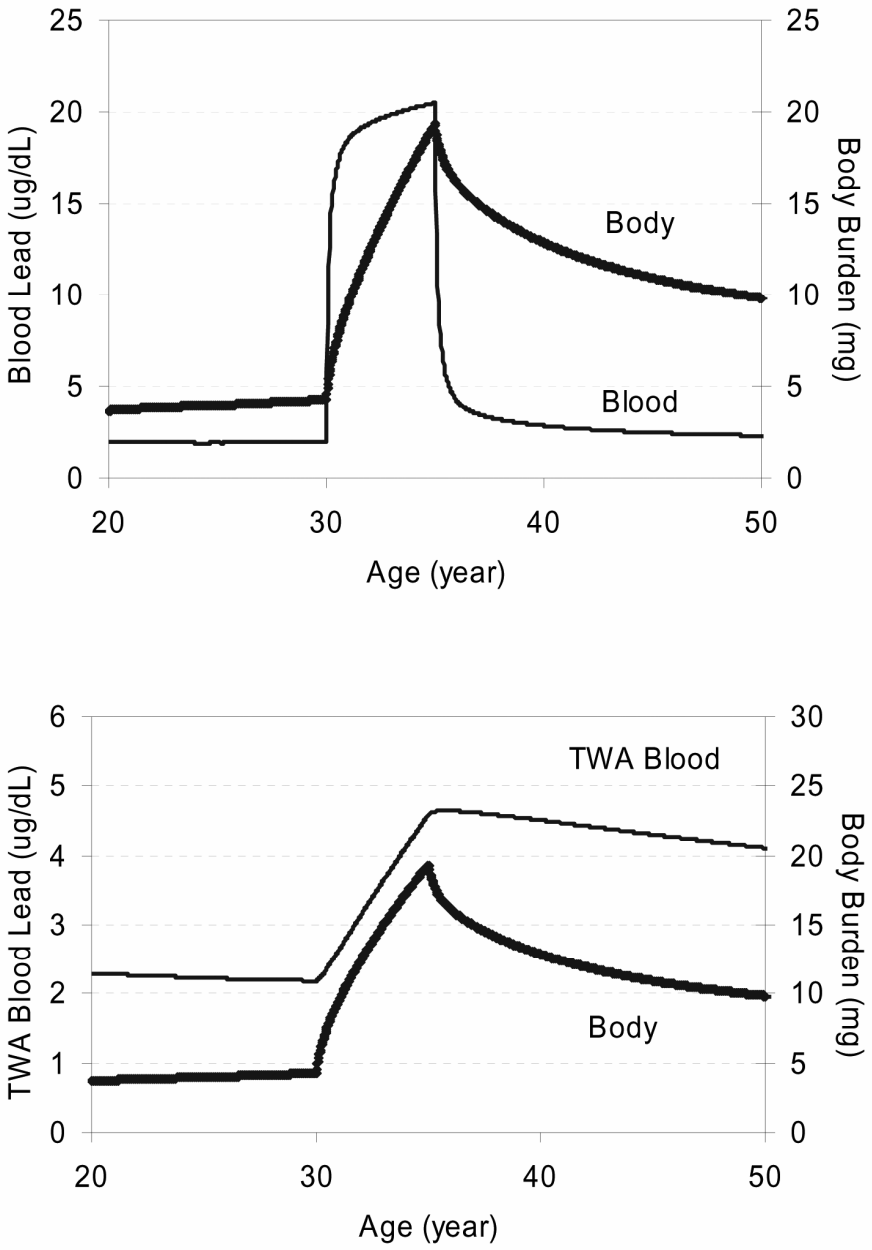

Figure 6-2.2. Simulation of relationship between blood lead concentration and body burden in adults. A constant baseline intake gives rise to a quasi-steady state blood lead concentration, while the body burden continues to increase, largely as a consequence of retention of lead in bone (upper panel). An abrupt change in lead uptake gives rise to a relatively rapid change in blood lead, to a new quasi-steady state, and a relatively small change in body burden. The long-term average blood lead concentration more closely tracks the pattern of change in body burden (lower panel). Simulation based on lead biokinetics model of Leggett (1993) .

5-year period of increased lead uptake; however, the blood lead concentration prior to and 1 year following cessation of the increased uptake has not changed (~2 μg/dL). Therefore, a single blood lead concentration measurement, or a series of measurements taken over a short-time span, can be expected to be a relatively poor index of lead body burden. On the other hand, an average of individual blood lead concentrations measured over a longer period of time (long-term average blood lead concentrations) can be expected to be a better index of body burden. In the hypothetical simulation shown in Figure 6-2.2, both the long-term average blood lead concentration and the body burden have approximately doubled.

The disparity in the kinetics of blood lead and body burden has important implications for the interpretation of blood lead concentration measurements in epidemiology studies. Cross-sectional studies, by design, sample blood lead concentration at one time or over relatively narrow windows of time. In these samples, the blood lead concentration may or may not reflect well the body burden; it is more likely to do so if the measured value is a reflection of the long-term average blood lead concentration. However, in cross-sectional samples, this cannot be ascertained. Longitudinal sampling provides a means for estimating average blood lead concentrations over time, and such estimates are more likely to be more strongly influenced by differences in body burden, than by differences in short-term variability in exposure. The degree to which repeated sampling will reflect the actual long-term time-weighted average blood lead concentration will depend on the sampling frequency in relation to variability in exposure. High frequency variability in exposures can produce episodic (or periodic) oscillations in blood lead concentration and body burden that may not be captured with low sampling frequencies.

The same basic concepts described above regarding lead biokinetics of adults also apply to children. The empirical basis for the understanding of the biokinetics of lead in children is much weaker than that for adults. However, based on the understanding of bone mineral kinetics and its importance as a mechanism for uptake and loss of lead from bone (Leggett, 1993; O'Flaherty, 1991, 1993, 1995), the slow pool, described above for adults, is thought to be much more labile in children, reflecting a more rapid turnover of bone mineral in children. As a result, changes in blood lead concentration in children are thought to more closely parallel changes in total body burden (Figure 6-2.3). Nevertheless, in children, as in adults, the long-term time-weighted average blood lead concentration is more likely to provide a better reflection of lead body burden than a single sample.

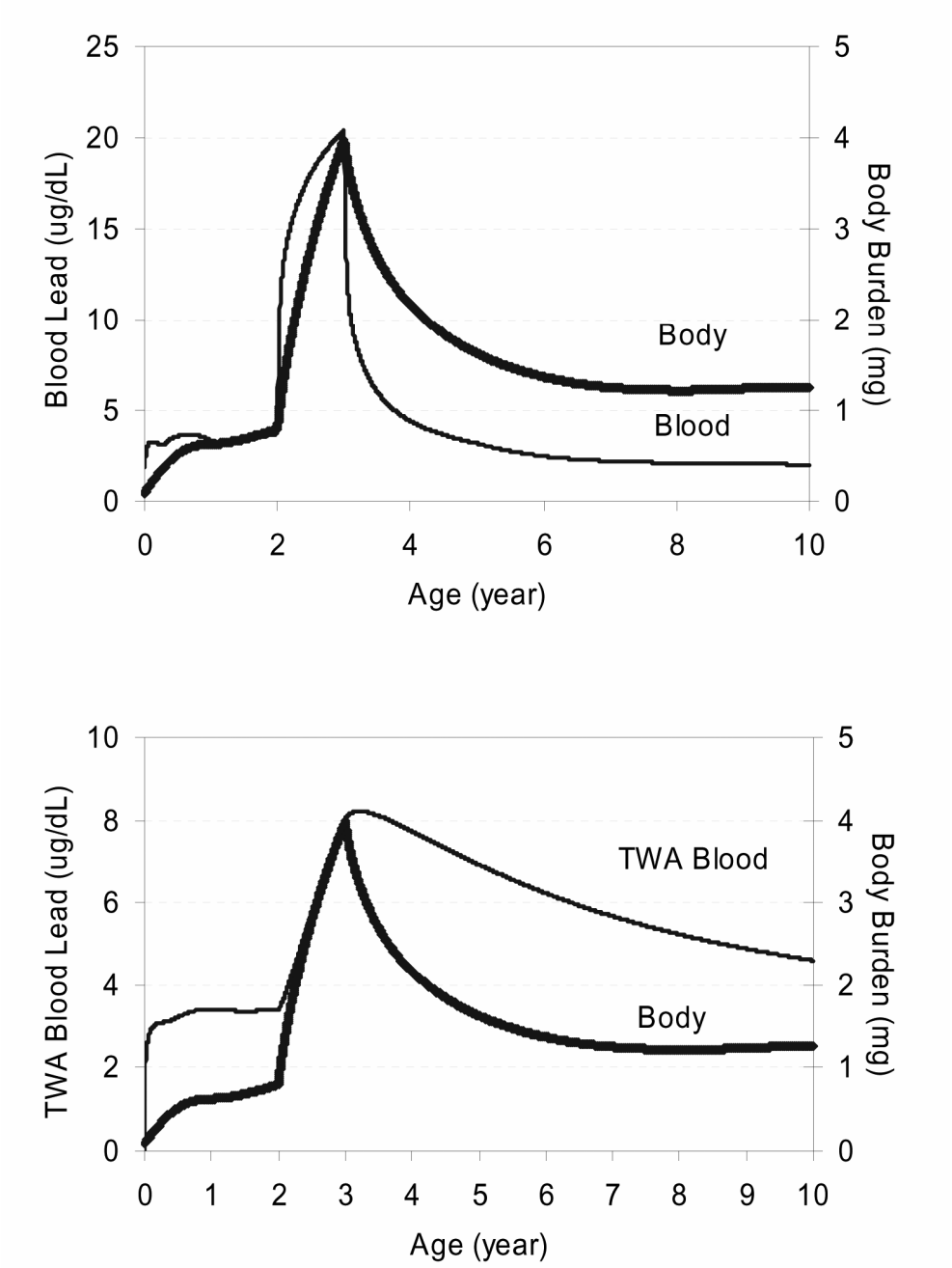

Figure 6-2.3. Simulation of relationship between blood lead concentration and body burden in children. Blood lead concentration is thought to parallel body burden more closely in children than in adults, due to more rapid turnover of bone and bone-lead stores in children (upper panel). Nevertheless, the long-term average blood lead concentration more closely tracks the pattern of change in body burden (lower panel). Simulation based on Leggett (1993) lead biokinetics model.

6.2.1.5 Blood Lead as a Biomarker of Lead Exposure

Characterizing quantitative relationships between external lead exposures and blood lead concentrations has become central to concentration-response analyses for human populations exposed to lead. The 1986 Lead AQCD summarized the empirical basis for this as it stood at the time. A summary of empirically-derived regression slope factors relating lead exposures and blood lead is provided in Abadin and Wheeler (1993). More recent meta-analyses, based on structure equation modeling, provide further support for quantitative relationships between lead exposures and blood lead concentrations in children (e.g., U.S. Environmental Protection Agency, 2001; Lanphear et al., 1998; Succop et al., 1998).

As noted above, the elimination half-life of lead in blood is ~25 to 30 days (Chamberlain et al., 1978; Rabinowitz et al., 1976; Griffin et al., 1975); therefore, the blood lead concentration mainly reflects the exposure history for the previous few months. However, a single blood lead measurement cannot distinguish between a history of long-term low level lead exposure or a history that includes higher acute exposures. This is illustrated in Figure 6-2.4. Two hypothetical children are simulated. Child A has a relatively constant lead intake from birth; whereas Child B has the same long-term lead intake as Child A, with a 1-year elevated intake which begins at age 24 months (Figure 6-2.4, upper panel). The absorption fraction is assumed to be the same for both children. Blood lead samples 1 and 5, or 2 and 4, will yield similar blood lead concentrations (~3 or 10 μg/dL, respectively), yet the exposure contexts for these samples are very different. Two samples (e.g., 1 and 2, or 4 and 5), at a minimum, are needed to ascertain if the blood lead concentration is changing over time. The rate of change can provide information about the magnitude of change in exposure, but not necessarily about the time history of the change (Figure 6-2.4, lower panel). Here again, time-integrated measurements of lead concentration may provide a means for accounting for some of these factors and, thereby, provide a better measure of long-term exposure. The same concepts apply to estimation of long-term exposure based on blood lead measurements in adults (Gerhardsson et al., 1992, 1995a; Roels et al., 1995).

An additional complication is that the relationship between lead intake and blood lead concentration is curvilinear; that is, the increment in blood lead concentration per unit of lead intake decreases with increasing blood lead concentration, both in children (Lacey et al., 1985; Ryu et al., 1983; Sherlock and Quinn, 1986) and in adults (Kehoe, 1987; Laxen et al., 1987;

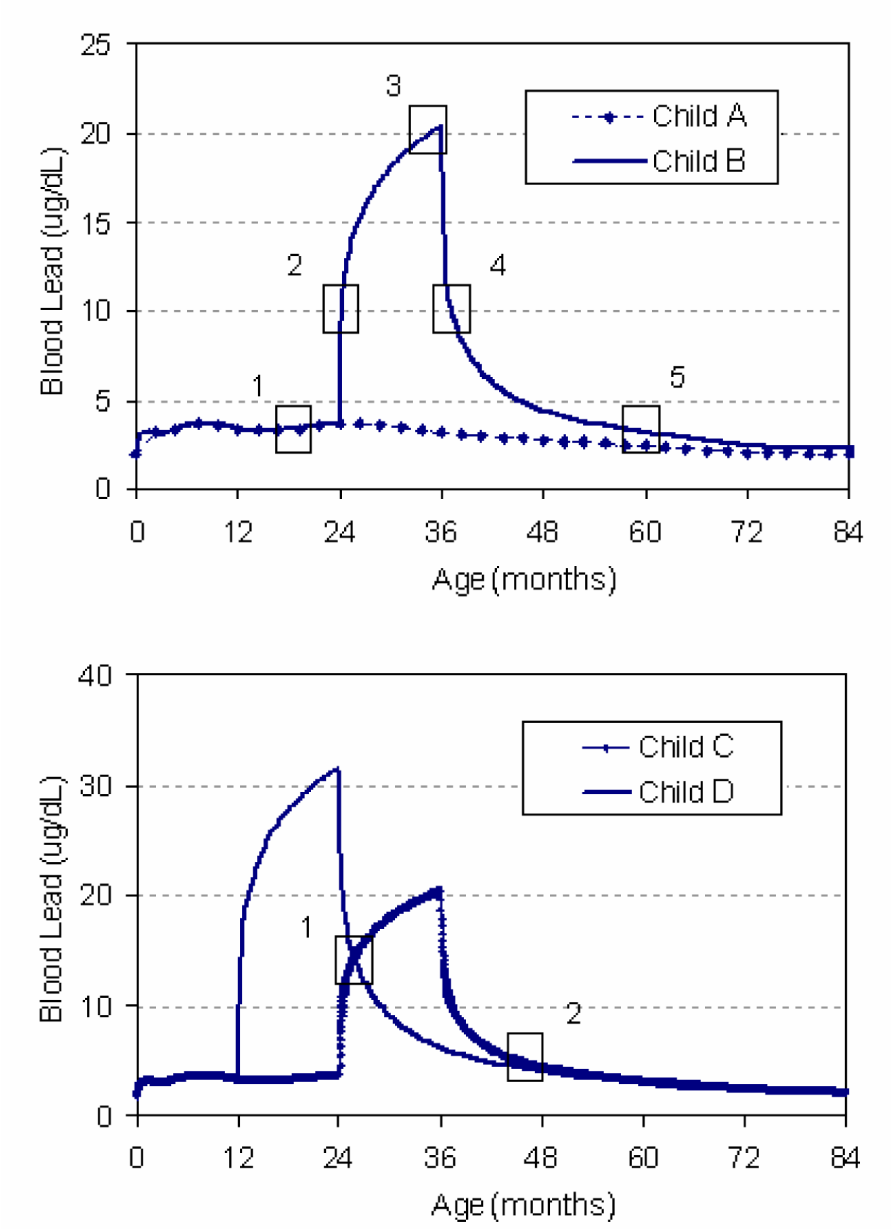

Figure 6-2.4. Simulation of temporal relationships between lead exposure and blood lead concentration in children. Child A and Child B have a relatively constant basal lead intake (µg/day/kg body weight) from birth; Child B experiences 1-year elevated intake which begins at age 24 months (upper panel). Blood lead samples 1 and 5, or 2 and 4, will yield similar blood lead concentrations (~3 or 10 µg/dL, respectively), yet the exposure scenarios for these samples are very different. As shown in the example of Child C and Child D, two samples can provide information about the magnitude of change in exposure, but not necessarily the temporal history of the change (lower panel).

Pocock et al., 1983; Sherlock et al., 1982, 1984). The nonlinearity is evident even at blood lead concentrations below 25 µg/dL (Figure 6-2.5). The nonlinearity in the lead intake-blood lead concentration relationship is derived, at least in part, from a capacity limitation in the accumulation of lead in erythrocytes (Bergdahl et al., 1997, 1998, 1999; Manton et al., 2001; Smith et al., 2002). A capacity-limited process may also reside at the level of intestinal absorption; however, the dose at which absorption becomes appreciably limited in humans is not known. Lead intake-blood lead relationships also vary (a) with age, as a result of age-dependency of gastrointestinal absorption of lead, and (b) with diet and nutritional status (Mushak, 1991).

The blood lead concentration is also influenced by lead in bone. Evidence for the exchange of bone lead and soft tissue lead stores comes from analyses of stable lead isotope signatures of lead in bone and blood. As noted earlier, bone lead likely contributes to the slow phase of elimination of lead from blood that has been observed in retired lead workers (Christoffersson et al., 1986; Schütz et al., 1987). Bone lead stores may contribute 40-70% of the lead in blood (Smith et al., 1996). This contribution increases during pregnancy, when mobilization of bone lead increases, apparently as the bone is resorbed to produce the fetal skeleton (Gulson et al., 2003). The mobilization of bone lead during pregnancy may contribute, along with other mechanisms (e.g., increased absorption), to the increase in lead concentration that has been observed during the later stages of pregnancy (Gulson et al., 1997; Lagerkvist et al., 1996; Schuhmacher et al., 1996). In addition to pregnancy, other states of increased bone resorption appear to result in release of bone lead to blood; these include lactation, osteoporosis, and menopause (Gulson et al., 2003). These observations are consistent with epidemiologic studies that have shown increases in blood lead concentration after menopause and in association with decreasing bone density in postmenopausal women (Hernandez-Avila et al., 2000; Nash et al., 2004; Symanski and Hertz-Picciotto, 1995). The relationship between blood and bone lead is discussed further in Section 6.2.2 on bone lead as a biomarker of lead exposure.

6.2.1.6 Summary of Blood Lead as a Biomarker of Lead Body Burden and Exposure

The blood lead concentration measured in an individual will be determined by the recent exposure history of the individual, as well as the long-term exposure history that gives rise to accumulated bone lead stores. The contribution of the latter to blood lead may change with the

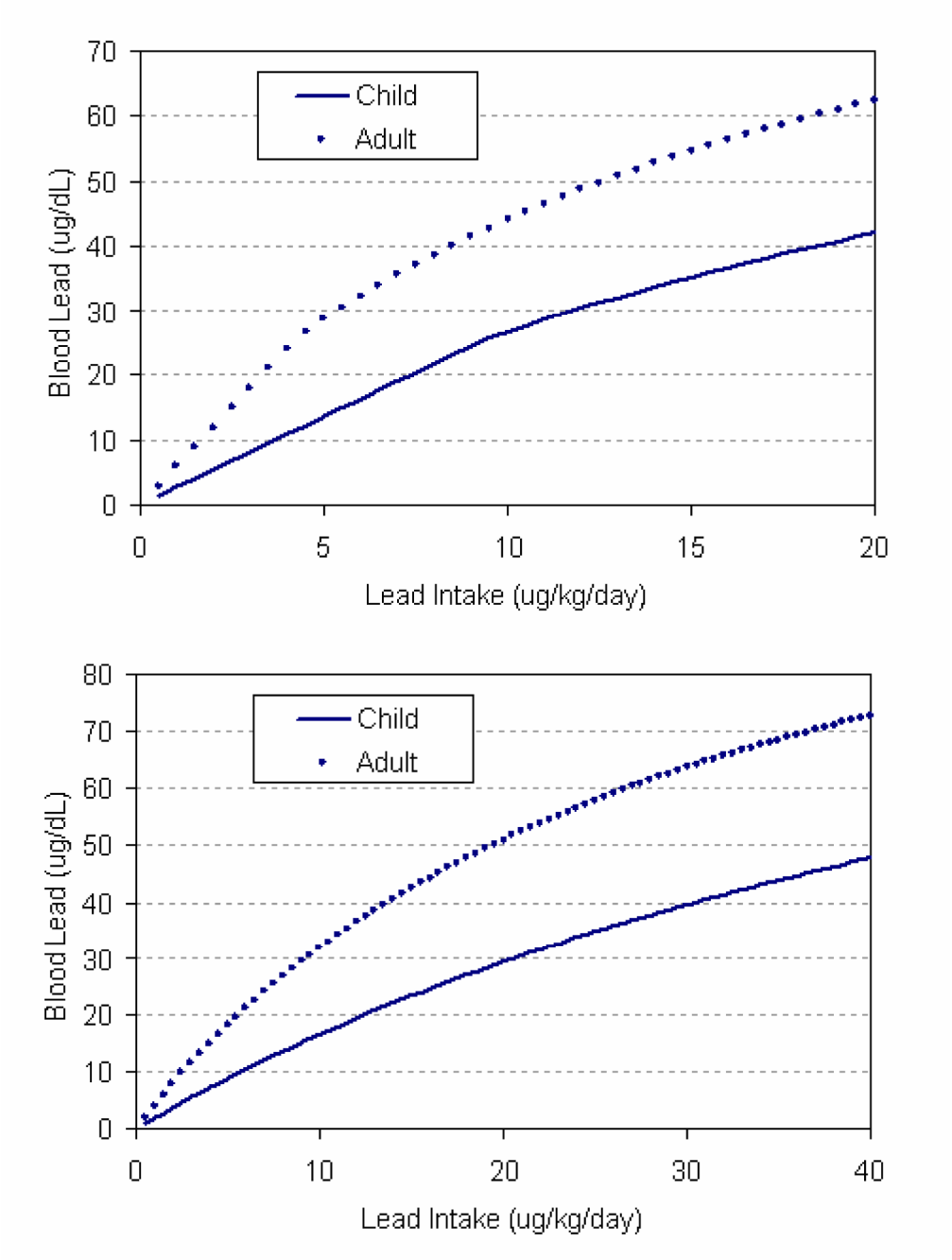

Figure 6-2.5. Simulation of relationships between lead intake and blood lead concentration in adults and children. The relationship between lead intake and blood lead concentration is curvilinear in adults and children. Predictions are for a 2-year-old child and 30-year-old adult, for a constant lead intake (µg/kg/day). Predictions are based on Leggett (1993, upper panel) and O'Flaherty (1993, 1995, lower panel).

duration and intensity of the exposure, age, and various physiological variables (e.g., nutritional status, pregnancy, menopause). Longitudinal measurements of blood lead can be expected to provide a more reliable measure of exposure history of an individual (and will more closely parallel body burden) compared to a single measurement; however, the degree to which this will apply will depend on the sampling frequency with respect to the temporal pattern of exposure.

In general, higher blood lead concentrations can be interpreted as indicating higher exposures (or lead uptakes); however, they do not necessarily predict appreciably higher body burdens. Similar blood lead concentrations in two individuals (or populations) do not necessarily translate to similar body burdens or similar exposure histories.

6.2.2 Lead in Bone

6.2.2.1 Summary of Key Findings from the 1986 Lead AQCD

In the 1986 Lead AQCD, the discussion on the distribution of lead in bone was fairly limited and mostly based on postmortem studies. The distribution between the two major compartments of cortical and trabecular bone were addressed especially based on the pioneering isotopic work of Rabinowitz et al. (1977). Estimates of the amount of lead in bone were also provided. There was limited discussion of the half-life of lead in bone as being on the order of several decades.

One of the major conclusions of the 1986 Lead AQCD regarding bone lead was that the traditional view that the skeletal system was a total sink for body lead was now giving way to the notion that there were at least several bone compartments for lead, with different mobility profiles. The possibility of bone lead serving as a source of long-term internal exposure was also considered.

Since 1986, the main focus of lead in bone studies has been on occupationally-exposed subjects, because of concern, until more recent times, about the ability to measure lower levels of lead in bone from environmentally-exposed subjects. Furthermore most of the focus has been on adult males, with very few studies on females and children. The newly available studies of lead in bone are discussed in the following sections.

6.2.2.2 Methodology of Bone Lead Analysis

6.2.2.2.1 Analytical Methods for Measuring Lead in Bone

Bone is comprised of two main types (cortical and trabecular) that have distinct rates of turnover and lead release, resulting in potential differences in implications with respect to toxicity aspects (further discussed in Section 6.2.2.3). The most commonly measured bones are the tibia, calcaneus, patella, and finger bone. For cortical bone, the midpoint of the tibia is measured. For trabecular bone, both the patella and calcaneus are measured. Recent studies favor measurement of the patella, because it has more bone mass and may afford better measurement precision than the calcaneus. The advantages and disadvantages of patella and calcaneus sites have not been thoroughly investigated. Bone lead measurements in cadavers, environmentally-exposed subjects, and occupationally-exposed subjects are presented in Annex Tables AX6-2.3, AX6-2.4, and AX6-2.5, respectively.

Bone analysis methods for in vivo measurements have included AAS, ASV, ICP-AES, ICP-MS, laser ablation inductively coupled plasma mass spectrometry (LAA ICP-MS), thermal ionization mass spectrometry (TIMS), synchrotron radiation induced X-ray emission (SRIXE), particle induced X-ray emission (PIXE), and X-ray fluorescence (XRF). Since the 1986 Lead AQCD, there have been many new papers published on bone lead using XRF. The upsurge in popularity of the XRF method has paralleled a decline in the use of the other methods.

In the past, two main approaches for XRF measurements have been used to measure lead concentrations in bone, the K-shell and L-shell methods. The K-shell method is now the most widely used, as there have been no further developments in L-shell devices since the early 1990s. The K-shell methods using ^{57}Cd and ^{109}Cd have been described in detail by Somervaille et al. (1989). Briefly, the K-shell XRF method uses 88.034 keV gamma rays from ^{109}Cd to fluoresce the K-shell X-rays of lead.

Since 1986, several investigators have reported refinements to hardware and software to improve the precision and accuracy of XRF measurements and there have been a number of investigations into the precision, accuracy and variability in XRF measurements (e.g., Aro et al., 2000; Todd et al., 2000, 2001, 2002). Todd et al. (2000) provided a detailed discussion of factors that influence the variability and measurement uncertainty, including repositioning, sample measurement duration, overlying tissue, operator expertise, detector resolution, and changes to measurement process over time. Some of these aspects were also discussed by

Hu et al. (1995). From their cadaver and in vivo measurements, Todd et al. (2000) concluded that the uncertainty in an individual measurement was an underestimate of the standard deviation of replicate measurements, suggesting a methodological deficiency probably shared by most current ^{109}Cd-based K-shell XRF lead measurement systems. In examining the reproducibility of the bone lead measurements over a 4½ month period, Todd et al. found the average difference between the XRF results from short term and longer term measurements was 1.2 μg/g, indicating only a small amount of variability in the XRF results over a sustained period of time.

6.2.2.2.2 Statistical Methods for Analyzing Bone Lead Concentrations in Epidemiologic Studies

In the literature, XRF bone data has typically been reported in two ways: one involving a methodological approach to assessing the minimum detection limit and the other termed an epidemiologic approach by Rosen and Pounds (1998). In the methodological approach, a minimum detection limit is defined using various methods, including two or three times the square root of the background counts; one, two, or three times the SD of the background; and two times the observed median error. This approach relies upon the minimum detection limit to define a quantitative estimate that is of sufficient precision to be included in the statistical analysis. The following are examples of methodological minimum detection limits for bone lead analyses. Bellinger et al. (1994) observed minimum detection limits, equivalent to the SD, of 5.4 μg/g for tibia and 9.2 μg/g for patella. Using twice the median observed error, Gerhardsson et al. (1993) observed minimum detection limits of 9.8 μg/g for tibia and 19.1 μg/g for calcaneus. For finger bone lead measurements, Christoffersson et al. (1986) observed a minimum detectable limit of 20 μg/g, which was equivalent to three times the square root of the background counts.

With the epidemiologic approach, to determine the minimum detection limit of an instrument all values are used (including negative values), which results in extremely low detection limits. Rosen and Pounds (1998) noted that this approach yields population bone lead averages that they considered artificially low and inconsistent with observations from many other earlier studies. However, not including values that are negative or below the detection limit, or assigning these values a fixed number for the statistical analysis is also of concern. To examine and compare the two methods used to analyze data at low levels of bone lead concentration,

Kim et al. (1995) performed serial measurements on phantoms containing spiked amounts of lead. The results demonstrated that the use of methodological minimum detection limits to recode low-level observations reduced the efficiency of the analysis and the ability to distinguish between the phantoms. Using the epidemiologic approach of retaining all point estimates of measured bone lead concentrations provided less bias and greater efficiency in comparing the mean or median levels of bone lead of different populations.

6.2.2.3 Bone Lead as a Biomarker of Lead Body Burden

6.2.2.3.1 Uptake of Lead in Bone

The dominant compartment for lead in the body is in bones. In human adults, more than 90% of the total body burden of lead is found in the bones, whereas bone lead accounts for ~70% of the body burden in children (Barry, 1975). Bone is comprised of two main types, cortical and trabecular. The tibia consists of more than 95% cortical bone, the calcaneus and patella comprise more than 95% trabecular bone, and finger bone is a mixed cortical and trabecular bone although the second phalanx is dominantly cortical. The cortical and trabecular bones have distinct rates of turnover and lead release, as well as potentially different associated toxicity implications (Hu et al., 1998). For example, adult tibia has a turnover rate of about 2% per year whereas trabecular bone has a turnover rate of more than 8% per year (Rabinowitz, 1991). The proportion of cortical to trabecular bone in the human body varies by age, but on average is about 80 to 20 (International Commission on Radiological Protection, 1973). Although not so important for certain types of measurements, the periosteum is of limited dimension and may reflect a bone compartment of more rapid deposition and turnover of lead than the other two types (Skerfving et al., 1993), which would also likely have implications for toxicity, especially for chelation therapy.

Much of the understanding of bone structure and metal deposition comes from studies of radioactive elements (e.g., International Commission on Radiological Protection, 1996). Durbin (1992, page 823) suggests that there is "an initial deposition of lead on anatomical bone surfaces with some skewing to the well nourished trabecular surfaces in red marrow, intense deposits at bone growth sites, and later on, a nearly diffuse labeling throughout the bone volume. For constant intake of lead during growth, it is expected that lead will be nearly uniformly distributed in the mineralized bone. Single or irregular intakes during growth are expected to result in

residual buried lines and hotspots superimposed on a relatively uniform diffuse concentration in bone mineral volume. . . For example, periosteal and subperiosteal lead deposits in the long bones, including those of the hands and feet, are likely to be greater than at many other sites, since bone growth continues at the periosteal surface while the endosteal surface is resorbed."

The importance of bone marrow was also stressed by Salmon et al. (1999), with a key factor affecting lead uptake into bone being the fraction of bone surface in trabecular and cortical bone adjacent to active bone marrow. The fraction of total marrow that is red and active decreases from 100% at birth to about 32% in adulthood (Cristy, 1981). Early lead uptake is greater in trabecular bone due to its larger surface area and higher metabolic rate. Of the total bone surface against red marrow, 76% is trabecular and 24% is cortical endosteal (Salmon et al., 1999). Bone marrow has much lower lead concentrations than bone matrix (Skerfving et al., 1983).

6.2.2.3.2 Half-Life of Lead in Bone

Estimates of the half-life of lead in trabecular bone are partly dependent on the tissue analyzed and the "purity" of the trabecular component (e.g., patella, calcaneus, and phalanx). Earlier estimates of the half-life of lead in trabecular bone ranged from 12 to 19 years (Bergdahl et al., 1998; Gerhardsson et al., 1993). For cortical bone, estimates for the half-life of lead were on the order of 13 to 27 years (Bergdahl et al., 1998; Gerhardsson et al., 1993; Rabinowitz, 1991).

With respect to half-lives in bone, recent K-shell XRF bone studies have indicated that earlier concepts of a constant rate of removal of lead from bone throughout adulthood assumed in models of human metabolism (Leggett, 1993; O'Flaherty, 1993) may be incorrect. In a study of active and retired smelter workers, Brito et al. (2001) suggested that people less than 40 years old had a shorter half-life for the release of lead from the tibia than those older than 40 years, 4.9 years (95% CI: 3.6, 7.8) compared to 13.8 years (95% CI: 9.7, 23.8), respectively. Also, they suggested that less intensely exposed subjects with a lifetime averaged blood lead of $\leq$25 µg/dL had a shorter half-life in the tibia (6.2 years [95% CI: 4.7, 9.0]) than those with a lifetime averaged blood lead >25 µg/dL (14.7 years [95% CI: 9.7, 29.9]).

Even by the end of the sixth decade, ~35 to 40% of skeletal mass consists of unremodelled first generation bone acquired during childhood and adolescence (International

Commission on Radiological Protection, 1973). This statement contrasts with that of O'Flaherty (1993) who suggested that because of the relatively short half-life of lead in the bones of children that much of the lead incorporated during active growth would not persist into adulthood. In a comparison of lead in tooth dentine and the tibia from young adults who were followed up after a period of 13 years, Kim et al. (1996) suggested that "pockets" of lead acquired in childhood may persist into adults. Likewise, McNeill et al. (2000) compared tibia lead levels and cumulative blood lead indices in a population of 19 to 29 year olds who had been highly exposed to lead in childhood from the Bunker Hill, Idaho smelter. They concluded that lead from exposure in early childhood had persisted in the bone matrix until adulthood.

6.2.2.3.3 Changes in Bone Lead Concentrations with Age

Conventional and XRF analyses of bone have shown significant increases in bone lead with age (Hu et al., 1990, 1996; Kosnett et al., 1994; Morgan et al., 1990). Kosnett et al. (1994) observed no significant change in bone lead concentrations up to age 20 years, but found an increasing trend with the same slope for men and women between the ages of 20 to 55 years and an increase to a faster rate in men older than 55 years. Kosnett et al. reanalyzed earlier cadaver cortical bone data of Drasch et al. (1987) and found that male bone lead values increased significantly after age 40 years, whereas female values slightly declined. A similar analysis of the post-mortem data of Barry (1975) showed an upward inflection for all males after age 35 years. Kosnett et al. (1994) found no significant slope to the relationship between age and bone lead for the 10 to 20 year old subjects, in contrast to Barry (1975) and Drasch et al. (1987).

Annual increments of lead to bone vary although no attempt has been made to determine whether the differences are significant. For example, the annual increment of 0.46 μg/g bone mineral/year found by Gordon et al. (1993) was slightly lower than that found by Somervaille et al. (1989), but the difference was not significant. After age 20 years, Kosnett et al. (1994) found the annual increment to be 0.38 μg/g bone mineral/year. Hu et al. (1990) reported a value of 0.31 μg/g bone mineral/year for subjects ranging in age from 20 to 58 years.

6.2.2.4 Distribution of Lead from Bone into Blood and Plasma

6.2.2.4.1 Contribution of Bone Lead to Blood Lead

Although the skeleton was recognized as a potentially significant contributor to blood lead in the 1986 Lead AQCD, there have been several investigations using both bone lead XRF and stable lead isotope methods which have helped quantify the contribution. The earlier estimation of skeletal contribution to blood lead was 70% by Manton (1985) and ~65% ranging up to 100% by Schütz et al. (1987). The more recent isotope studies confirmed these estimates. Using female immigrants to Australia and their children, Gulson et al. (1995, 1997, 1999a) found a mean value of 50% (range 16-73%) deriving from the skeleton. Smith et al. (1996) found a range of 40–70% in five patients who underwent total hip or knee joint replacement. Gwiazda et al. (2005) observed a range of 40-65% in two children and >90% in one child. Studies examining the bone lead contribution to blood lead are presented in Annex Table AX6-2.6.

The contribution of skeletal lead to blood lead was further examined in females from varying environments. In middle-aged to elderly subjects (46-74 years), an increase of 19 μg/g of lead in tibia bone mineral was associated with an increase in blood lead of 1.7 μg/dL, which corresponds to a 0.09 μg/dL increase in blood lead per 1 μg/g bone mineral (Korrick et al., 2002). A study of 108 former workers at the Bunker Hill smelter in northern Idaho and 99 referents from the Spokane, WA area examined the endogenous bone lead release rate of postmenopausal and premenopausal women (Popovic et al., 2005). The results indicated that the endogenous release rate in postmenopausal women (0.13 μg/dL per μg/g bone) was greater than the rate found in premenopausal women (0.07 μg/dL per μg/g bone). In a Mexico City study, the endogenous bone lead release rate in postmenopausal women also was observed to be double that in premenopausal women (Garrido-Latorre et al., 2003). A change of 10 μg/g bone mineral was associated with an increase in blood lead of 1.4 μg/dL in postmenopausal subjects, compared to an increase of 0.8 μg/dL in premenopausal women. Lactation was also found to affect the endogenous bone lead release rate. After adjusting for patella lead concentration, an increase in blood lead levels of 12.7% (95% CI: 6.2, 19.6) was observed for women who practiced partial lactation and an increase of 18.6% (95% CI: 7.1, 31.4) for women who practiced exclusive lactation compared to those who stopped lactation (Téllez-Rojo et al., 2002).

The mean cortical leads to current blood lead ratios for occupationally-exposed subjects are shown in Figure 6-2.6. Box plots were calculated using data from the following studies:

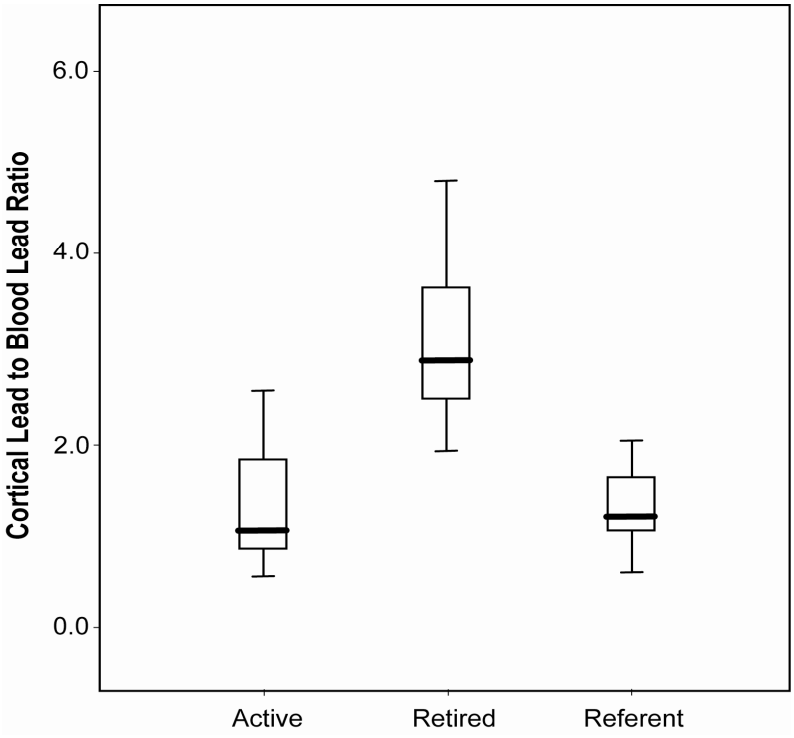

Figure 6-2.6. Cortical lead to blood leads ratios for occupationally-exposed subjects (both active and retired) and referents. Data compiled from several studies. See text for more details.

Bergdahl et al., 1998; Brito et al., 2002; Christoffersson et al., 1984; Erfurth et al., 2001; Erkkilä et al., 1992; Fleming et al., 1998; Gerhardsson et al., 1993; Hänninen et al., 1998; Juarez-Perez et al., 2004; Popovic et al., 2005; Roels et al., 1995; Schwartz et al., 2000a, 2000b; Somervaille et al., 1988, 1989; Todd et al., 2001. The mean cortical lead to current blood lead ratio is about 1.2 (range 0.4-2.6) for active employees (n = 17). For retired employees (n = 7), the mean is 3.2 (range 2.0-5.3), while for environmentally-exposed referent subjects from these industries (n = 7) the mean ratio is about 1.3 (range 1-2.2). The differences in the cortical lead to blood lead ratio between active and retired employees and retired employees and referents are significant ($p < 0.01$) but not between active employees and referents. Several investigators have pointed out the weak association between bone lead and blood lead in active employees in comparison with the stronger association with retired employees (e.g., Erkkilä et al., 1992; Fleming et al., 1997; Gerhardsson et al., 1993). This is likely because circulatory lead of active employees

reflects mainly ongoing exposure whereas that in retired employees is more dependent on lead released from the skeleton.

The mean tibia lead to current blood lead ratios for environmentally-exposed subjects is shown in Figure 6-2.7. The box plot for pregnancy-related subjects was calculated using data from the following studies: Brown et al., 2000; Chuang et al., 2001; Ettinger et al., 2004; Gomaa et al., 2002; Gonzalez-Cossio et al., 1997; Hernandez-Avila et al., 1996, 1998, 2002, 2003; Hu et al., 1996; Moline et al., 2000; Rothenberg et al., 2000; Sanin et al., 2001; Téllez-Rojo et al., 2002, 2004. The box plot for middle-aged and elderly subjects included the following studies: Berkowitz et al., 2004; Cheng et al., 1998a; Garrido-Lattore et al., 2003; Hu et al., 1996, 2001; Korrick et al., 2002; Kosnett et al., 1994; Oliveira et al., 2002; Schafer et al., 2005; Tsaih et al., 2004; Webber et al., 1995. The box plot for the younger subjects (age range 1-30 years) included Farias et al., 1998; Kim et al., 1996; Rosen et al., 1989; Stokes et al., 1998. The mean tibia lead to blood lead ratio for pregnancy-related subjects ($n = 21$) is 1.5 (range 1.0-4.2) and is statistically significantly different ($p < 0.001$) from the mean ratio of 3.4 (range 1.6-5.4) for middle-aged to elderly subjects ($n = 27$). Similar relationships are observed for the patella lead to blood lead ratios for pregnancy-related subjects and middle-aged to elderly subjects.

In several other studies of environmentally-exposed subjects, there is a stronger relationship between patella lead and blood lead than tibia lead and blood lead (e.g., Hernandez-Avila et al., 1996; Hu et al., 1996, 1998). Hu et al. (1998) suggest that these relationships indicate that trabecular bone is the predominant bone type providing lead back into circulation under steady-state and pathologic conditions.

6.2.2.4.2 Partitioning of Bone Lead into Plasma

Although most of the lead in whole blood is associated with erythrocytes (~99%), it has been suggested that the small fraction of lead in plasma (<0.3%) may be the more biologically labile and toxicologically active fraction of the circulating lead. Several authors have proposed that lead released from the skeleton was preferentially partitioned into serum compared with red cells (Cake et al., 1996; Hernandez-Avila et al., 1998; Tsaih et al., 1999) with one explanation being that the lead from endogenous sources was in a different form to that from exogenous sources. However, this concept has been withdrawn by its main proponents. In contrast to using

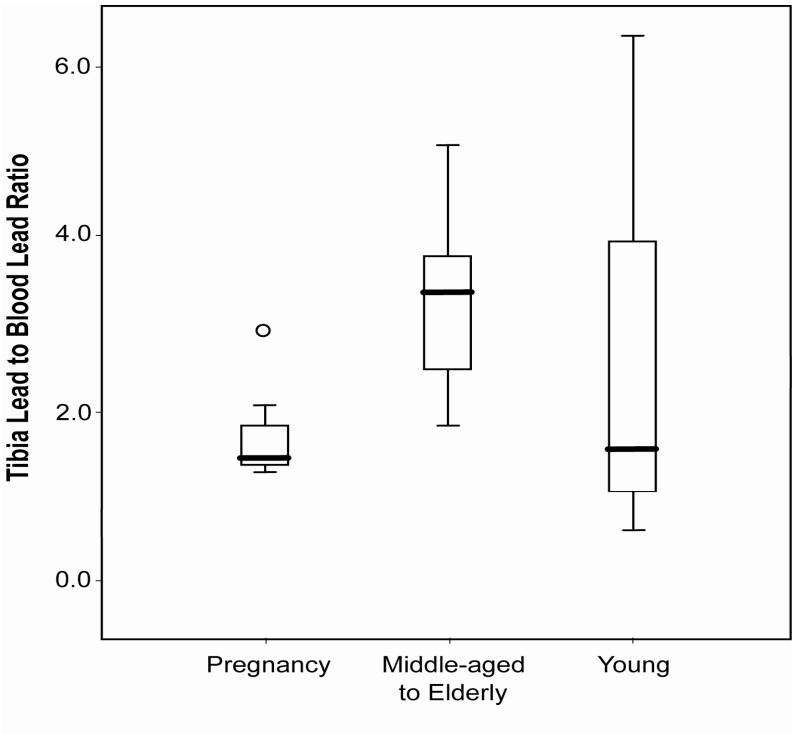

Figure 6-2.7. Tibia leads to blood lead ratios for environmentally-exposed pregnancy-related subjects, middle-aged to elderly subjects, and younger subjects. Data compiled from several studies. See text for more details.

urine as a proxy for serum and measuring lead isotopes, Gulson et al. (2000) concluded that there was no evidence for preferential partitioning of lead into serum compared with whole blood.

6.2.2.5 Mobilization of Lead From Bone

Although earlier investigators such as Brown and Tompsett (1945), Ahlgren et al. (1976) and Christoffersson et al. (1984) suggested that the skeleton was a potential endogenous source of lead poisoning, the opposing concept of the skeleton as a "safe" repository for lead persisted until the mid-1980s and early 1990s. Potential mobilization of lead from the skeleton could occur at times of physiological stress associated with enhanced bone remodeling such as during pregnancy and lactation (Hertz-Picciotto et al., 2000; Manton, 1985; Silbergeld, 1991), menopause or in the elderly (Silbergeld, 1991; Silbergeld et al., 1988), extended bed rest (Markowitz and Weinberger, 1990), hyperparathyroidism (Kessler et al., 1999), and weightlessness. The lead deposited in the bone of adults can serve to maintain blood lead levels

long after exposure has ended (Fleming et al., 1997; Gulson et al., 1995; Inskip et al., 1996; Kehoe, 1987; Manton, 1985; Nilsson et al., 1991; O'Flaherty et al., 1982; Schütz et al., 1987; Smith et al., 1996).

In the 1986 Lead AQCD, there was a comprehensive summary of chelation therapies and the recognition that there was limited release of lead from bones. The potential role of bone lead as an endogenous source of lead in blood, resulting in elevated levels for former lead employees, was mentioned although data to support this hypothesis were limited.

6.2.2.5.1 Mobilization of Lead from Bone during Pregnancy and Lactation

Bone lead studies of pregnant and lactating subjects are summarized in Annex Table AX6-2.7. Most of the bone XRF studies on pregnancy and lactation have focused on subjects from Mexico City and Latin subjects from Los Angeles, California. Relationships and/or health outcomes from these investigations include: patella bone as a significant contributor to blood lead (Brown et al., 2000; Hernandez-Avila et al., 1996); a positive association between plasma lead and bone lead in the highest bone lead group of pregnant women (Téllez-Rojo et al., 2004); a positive association of tibia and calcaneus lead with prenatal lead concentration, and calcaneus lead with postnatal lead (Rothenberg et al., 2000); a positive association of tibia lead and seasonal variations in blood lead (Rothenberg et al., 2001); maternal tibia and patella lead as significant predictors of fetal exposure determined using cord blood (Chuang et al., 2001); a positive association of calcaneus lead and increased systolic and diastolic blood pressure in the third trimester (Rothenberg et al., 2002); an inverse relationship between maternal tibia and patella lead, and birth weight (Gonzalez-Cossio et al., 1997; Sanin et al., 2001); an inverse association between tibia lead and birth length, and patella lead and head circumference (Hernandez-Avila et al., 2002); an inverse association of maternal patella bone and Mental Development Index (Gomaa et al., 2002); increased bone resorption during lactation (Téllez-Rojo et al., 2002); increased lead in breast milk with an increase in patella and tibia lead (Ettinger et al., 2004).

Lead isotope studies on immigrant women to Australia (Gulson et al., 1997, 1998a) confirmed the earlier work of Manton (1985) of increased blood lead during pregnancy. Gulson et al. reported that, during pregnancy, blood lead concentrations in the first immigrant cohort (n = 15) increased by an average of about 20% compared to non-pregnant migrant controls

(n = 7). The percentage change in blood lead concentration was significantly greater during the postpregnancy period than during the second and third trimesters ($p < 0.001$). Skeletal contribution to blood lead, based on the isotopic composition for the immigrant subjects, increased in an approximately linear manner during pregnancy. The mean increases for each individual during pregnancy varied from 26% to 99%. Skeletal lead contribution to blood lead was significantly greater during the postpregnancy period than during the second and third trimesters. The contribution of skeletal lead to blood lead during the postpregnancy period remained essentially constant at the increased level of lead mobilization. In a follow-up study using a different immigrant cohort of 12 women with calcium supplementation at the recommended level of approximately 1,000 mg/day (National Institutes of Health, 1994), Gulson et al. (2004) found increased mobilization of lead occurred in the third trimester rather than in the second trimester as observed with first cohort. In addition, the extra flux released from bone during late pregnancy and postpartum varied from 50 to 380 μg (geometric mean 145 μg) compared with 330 μg in the previous cohort.

In an extended monitoring of 7 subjects for up to 22 months postpartum, Gulson et al. (1999a) found that blood lead concentrations in some of the subjects decreased to about half the earlier levels almost immediately after cessation of breastfeeding. However, in 4 of the 7 cases there was a rebound in blood lead concentrations that exceeded the earlier levels in 3 cases. The authors interpreted these results to indicate that there is ongoing increased mobilization of lead from the maternal skeleton for much longer than predicted, probably associated with remodeling processes. Also using lead isotopes, Manton et al. (2003) observed that blood lead concentrations decreased in early pregnancy and rose during late pregnancy. They attributed these results to changes in bone resorption with decoupling of trabecular and cortical bone sites.

6.2.2.5.2 Transplacental Transfer of Lead and Transfer through Breast Milk

Transplacental transfer of lead in humans has been suggested in a number of studies based on cord blood to maternal blood lead ratios ranging from about 0.6 to 1.0 at the time of delivery. Maternal-to-fetal transfer of lead appears to be related partly to the mobilization of lead from the maternal skeleton. Evidence for transfer of maternal bone lead to the fetus has been provided from stable lead isotope studies in cynomolgus monkeys (*Macaca fascicularis*). Approximately 7 to 39% of the maternal lead burden that is transferred to the fetus appears to derive from the

maternal skeleton (Franklin et al., 1997; O'Flaherty et al., 1998). Further evidence for maternal-to-fetal transfer of lead in humans can be gained from stable lead isotope measurements. For example, a 0.99 correlation in lead isotopic ratios for maternal and cord blood (Manton, 1985; Gulson et al., 1998b) and the similarity of isotopic ratios in maternal blood and in blood and urine of newly-born infants provide strong evidence for placental transfer of lead to the fetus (Gulson et al., 1999b).

Breast milk can also be a pathway of maternal excretion of lead. However, given the very low lead concentrations and analytical difficulties arising from high fat contents in breast milk, their analyses require careful attention. Selected studies appear to show a linear relationship between breast milk and maternal whole blood with the percentage of lead in breast milk compared with whole blood of <3% in subjects for blood lead concentrations ranging from 2 to 34 μg/dL. Blood lead concentrations in breastfed newborn infants decreased in spite of the maternal blood lead concentrations having risen or remained elevated postpartum compared to lower levels during prepregnancy or in the first trimester (Gulson et al., 1999b). Similar trends were noted by Manton et al. (2000). However, in a Mexico City study, an association between patella lead and blood lead concentrations was higher for women with partial lactation than for those who stopped lactation, and it was increased among women who breastfed exclusively (Téllez-Rojo et al., 2002). In another Mexico City study, Ettinger et al. (2004) concluded that an interquartile increase in patella lead was associated with a 14% increase in breast milk lead, whereas for tibial lead the increase was ~5%.

In conclusion, there is evidence that maternal-to-fetal transfer of lead occurs, likely resulting from the mobilization of lead from the maternal skeleton during pregnancy. Breast-fed infants appear to be at greater risk only if the mother is exposed to high lead concentrations either from exogenous sources or endogenous sources such as the skeleton.

6.2.2.5.3 Mobilization of Lead in Bone During Menopause and in the Elderly

Increases in blood lead for postmenopausal women have been attributed to release of lead from the skeleton associated with increased bone remodeling during menopause. Many of the studies have been based on blood lead concentration. Bone lead studies of menopausal and middle-aged to elderly subjects are summarized in Annex Table AX6-2.8.

Overall, the various studies of bone and blood lead levels, as well as hormone replacement therapy, have provided conflicting outcomes. Hormone replacement therapy alone or combined with calcium supplementation prevents bone resorption and increases the bone mineral density in trabecular and cortical bones of women with or without metabolic bone disease. The effect of hormone replacement therapy may result in a decrease of lead mobilization from bone along with a reduction in blood lead concentration levels. Several studies have found that tibia bone lead levels were higher in women who used hormone replacement therapy (Popovic et al., 2005; Webber et al., 1995). In contrast, other investigators have found no association between bone lead and use of estrogens (Berkowitz et al., 2004; Korrick et al., 2002). In addition, some studies observed a decrease in blood lead concentrations associated with hormone replacement therapy (Garrido-Latorre et al., 2003), whereas others observed no association (Webber et al., 1995).

The endogenous release rate of lead from bone in postmenopausal women was double the rate in premenopausal former smelter employees (Popovic et al., 2005) and environmentally-exposed women from Mexico (Garrido-Latorre et al., 2003). In middle-aged to elderly males from the Normative Aging Study, patella lead accounted for the dominant portion of variance in blood lead (Hu et al., 1996).

6.2.2.5.4 Effect of Nutritional Status on Mobilization of Lead from Bone

Most studies that investigated the effect of nutritional status on the mobilization of lead from the skeleton have examined the effects of calcium supplementation. Several studies have suggested that dietary calcium may have a protective role against lead by decreasing absorption of lead in the gastrointestinal tract and by decreasing the mobilization of lead from bone stores to blood, especially during periods of high metabolic activity of the bone such as pregnancy, lactation, and menopause. An inverse association between patella lead and low calcium intake in postpartum women has been found (Hernandez-Avila et al., 1996). In contrast, Rothenberg et al. (2000) observed that dietary calcium intake had no effect on calcaneus lead in women monitored during the third trimester and 1 to 2 months postpartum. Likewise, no effect from calcium supplementation on bone lead was found amongst lactating women from Mexico City (Téllez-Rojo et al., 2002), although in a follow-up study, Hernandez-Avila et al. (2003) reported a 16.4% decrease in blood lead concentration among women with the highest patella bone lead levels who

were taking supplements. Gulson et al. (2004) observed that calcium supplementation was found to delay increased mobilization of lead from bone during pregnancy and halved the flux of lead release from bone during late pregnancy and postpartum. In another study, women whose daily calcium intake was 850 mg per day showed lower amounts of bone resorption during late pregnancy and postpartum than those whose intake was 560 mg calcium per day (Manton et al., 2003). Téllez-Rojo et al. (2004) observed that plasma lead levels were inversely related to dietary calcium intake. Results for whole blood lead were similar but less pronounced.

Some researchers have noted concerns regarding potential lead toxicity resulting from calcium supplementation. However, Gulson et al. (2001) observed that lead in calcium or vitamin supplements did not appear to increase blood lead concentrations. No information was available on the effects of other nutritional supplements (e.g., iron or zinc) on lead body burden.

6.2.2.6 Summary of Bone Lead as a Biomarker of Lead Body Burden and Exposure

Bone accounts for more than 90% of the total body burden of lead in adults and 70% in children. In addition, the longer half-life of lead in bone, which largely depends on the bone type but is generally estimated in terms of years compared to days for blood lead, allows a more cumulative measure of lead dose. The more widespread use of in vivo XRF lead measurements in bone and indirect measurements of bone processes with stable lead isotopes since the 1986 Lead AQCD have enhanced the use of bone lead as a biomarker of lead body burden.

In addition to considering bone lead as an indicator of cumulative lead exposure, lead in the skeleton can also be regarded as a source of lead. Key studies have examined the contribution of bone lead to blood lead; the preferential partitioning of bone lead into plasma; mobilization of lead from bones during pregnancy, lactation, and menopause; and the role of nutritional supplementation in bone mobilization.

6.2.3 Lead in Teeth

6.2.3.1 Summary of Key Findings from the 1986 Lead AQCD

The importance of dentine as a potential indicator of lead exposure was noted in the 1986 Lead AQCD. There was more emphasis and optimism on using dentine to assess lead exposure in this document as the bone XRF method was in its infancy. The issue of deciduous tooth type was addressed but there was little information on permanent teeth. The portion of the tooth

analyzed (i.e., whole tooth or circumpulpal dentine) was also addressed. In the 1990 Addendum, the use of tooth lead as an exposure metric was described in a number of the longitudinal and cross-sectional studies.

6.2.3.2 Analytical Methods for Measuring Lead in Teeth

Analytical methods for tooth analysis vary from the most widely used AAS, to energy-dispersive XRF, laser ablation inductively coupled plasma mass spectrometry (LA-ICP-MS), and high precision lead isotopes.

As a standard analytical method has yet to be established for tooth lead analysis, some of the discrepancies in findings between studies could arise from several factors, including differences in tooth type, part of the tooth analyzed, and tooth location. Any real differences among populations are unlikely to be the result of physiological factors such as blood supply to teeth or mineralization rates. As enamel and dentine in different teeth calcify at overlapping but different times (Orban, 1953), they could retain varying amounts of lead.

In a systematic evaluation of the magnitude of random errors associated with dentine lead measurements, Fergusson et al. (1989) measured lead concentrations in two samples of dentine from 996 New Zealand children. They estimated that 15 to 20% of the variance was unexplained. Tests of differences of means and variances showed no significant differences between the two samples.

Lead measurements in deciduous teeth in individuals from urban and remote environments and from polluted environments are presented in Annex Tables AX6-2.9 and AX6 2.10, respectively. Based on the limited number of studies, it would appear that the range in whole deciduous tooth lead for environmentally exposed subjects is about 1–10 μg/g, but the most likely levels are <5 μg/g and probably even <2 μg/g. Studies of whole deciduous teeth from industrial environments, including those in urban settings, are also commonly much less than 10 μg/g.

The utility of circumpulpal dentine (Shapiro et al., 1973) as the metric of lead exposure in deciduous teeth has not been enthusiastically received. This is likely due to the separation difficulties, as well as the limited amount of circumpulpal dentine that may be present when the teeth are resorbed, prior to exfoliation.

In another approach to gain more information about exposure during pregnancy and early childhood, the teeth may be sectioned into dominantly enamel or dominantly dentine. These samples can then be analyzed for lead isotopic ratios and lead concentrations (Gulson and Wilson, 1994). Even for children living in lead mining and smelting communities, levels of lead in the enamel are generally low (<5 μg/g) and are consistent with other studies of whole teeth. However, higher levels are observed in the dentine samples (e.g., 32 μg/g), which likely reflect the early childhood exposure. Permanent teeth tend to have up to three times the level of lead compared with deciduous teeth, but the number of studies is very limited.

6.2.3.3 Tooth Lead as a Biomarker of Lead Body Burden

Compared with the amount of lead in the skeleton, tooth lead is a minor contributor to the body burden of lead. Most of the tooth lead information is based on analyses of deciduous teeth. There is still controversy over the amounts of lead in different whole teeth but it appears that the highest concentrations are in central incisors, with decreasing amounts in lateral incisors, canines, first molars, and second molars. Teeth from the upper jaw tend to have higher lead concentrations than those from the lower jaw.

As teeth accumulate lead, tooth lead levels are generally considered an estimate of cumulative lead exposure. Rabinowitz et al. (1993) found that tooth lead was a better measure of exposure than current blood lead levels; however, it was not a good measure of the child's cumulative exposure from birth to exfoliation due to the mobilization of lead from dentine.

Teeth are composed of several tissues formed over the years. Therefore, if a child's lead exposure during the years of tooth formation varied widely, different amounts of lead would be deposited at different rates (Rabinowitz et al., 1993). This may allow investigators to elucidate the history of lead exposure in a child.

Gulson and Wilson (1994) advocated the use of sections of enamel and dentine to obtain additional information compared with analysis of the whole tooth (e.g., Fosse et al., 1995; Tvinnereim et al., 1997). For example, deciduous teeth lead in the enamel provides information about in utero exposure whereas that in dentine from the same tooth provides information about postnatal exposure until the tooth exfoliates at about 6 to 7 years of age.

6.2.3.4 Relationship between Tooth Lead and Blood Lead

As with bone lead-blood lead relationships, there is interest in understanding more about potential relationships between tooth lead and blood lead. The tooth lead-blood lead relationship is more complex than the bone lead-blood lead relationship because of differences in tooth type, location, and analytical method.

Rabinowitz (1995) used studies which reported values for dentine, whole shed teeth, or crowns, but discarded those measuring circumpulpal dentine because of the higher values in this medium. The mean tooth lead levels varied from 2.8 to 12.7 μg/g and blood lead levels from 6.5 to 17 μg/dL. In a plot of blood versus tooth lead, Rabinowitz found a good fit ($R^2 = 0.97$; $p < 0.0001$) with the relationship:

$$\text{Tooth Lead } (\mu g/g) = \beta \times [\text{Blood Lead } (\mu g/dL)], \text{ where } \beta = 0.49 \text{ (SE } 0.04).$$

In an earlier Boston study, Rabinowitz et al. (1989) found that the association between tooth and blood lead increased with age, first achieving statistical significance at 18 months; by 57 months, the correlation coefficient was 0.56. A correlation of 0.47 was found between current blood lead and incisors amongst 302 German children (Ewers et al., 1982).

6.2.3.5 Mobilization of Lead from Teeth

Although mobilization of lead from bone appears well established, this is not the case for lead in teeth. Conventional wisdom has lead fixed once it enters the tooth. Although that may be the case for the bulk of enamel, it is not true for the surface of the enamel and dentine.

In evaluating deciduous teeth data, Rabinowitz et al. (1993) suggested that their data were compatible with a model that allows lead to be slowly removed from dentine. Such a process may be associated with resorption of the root and dentine that precedes exfoliation, which allows reequilibration of dentine lead with blood lead.

In children exposed to lead sources from mining, paint, or petrol in communities such as the Broken Hill lead mining community, Gulson and Wilson (1994) and Gulson (1996) showed that the source of lead from the incisal (enamel) sections was different from the source of lead in the cervical (dentine) sections of deciduous teeth, reflecting the change in lead from in utero exposure to early childhood. Based on changes in the isotopic composition of enamel and

dentine in deciduous teeth sections from the Broken Hill mining community children, Gulson (1996) estimated that lead is added to dentine at a rate of approximately 2-3% per year.

Stable lead isotopes and lead concentrations were measured in the enamel and dentine of permanent (n = 37) and deciduous teeth (n = 14) from 47 European immigrants to Australia to determine whether lead exchange occurs in teeth and how it relates to lead exchange in bone (Gulson et al., 1997). The authors concluded that enamel exhibited no exchange of its European-origin lead with lead from the Australian environment, whereas dentine lead exchanged with Australian lead to the extent of ~1 ± 0.3% per year.

6.2.3.6 Summary of Tooth Lead as a Biomarker of Lead Body Burden and Exposure

Tooth lead is a minor contributor to the total body burden of lead. Moderate-to-high correlations have been observed between tooth lead levels and blood lead levels. Differences in tooth type, part of the tooth analyzed, and tooth location may contribute to some of the discrepancies in findings between studies of tooth lead. As teeth are composed of several tissues formed over the years, if a child's lead exposure during the years of tooth formation varied widely, different amounts of lead would be deposited at different rates. Deciduous teeth lead in the enamel provides information about in utero exposure, whereas that in dentine provides information about postnatal exposure until the tooth exfoliates.

6.2.4 Lead in Urine

6.2.4.1 Summary of Key Findings from the 1986 Lead AQCD

The 1986 Lead AQCD provided an extensive discussion of the physiological basis for "chelatable" urinary lead. Also discussed was lead excretion provoked by EDTA, including the pools of lead in the body that might be mobilized in the EDTA provocation test, and the relationship between the outcome and blood lead concentration. The 1986 Lead AQCD noted observations that formed the basis for application of the EDTA provocation test for detecting elevated lead body burden.

6.2.4.2 Analytical Methods for Measuring Lead in Urine

Standard methods that have been reported for urine lead analysis are summarized in Annex Table AX6-2.1 and are, in general, the same as those analyses noted for determination of

lead in blood. Reported detection limits are approximately 50 µg/L for AAS, 5–10 µg/L for ICP-AES, and 4 µg/L for ASV for urine lead analyses. Sample preparation usually consists of wet ashing; however, chelation and solvent extraction has also been reported (National Institute for Occupational Safety and Health, 1994, 1977a).

6.2.4.3 Levels of Lead in Urine

A summary of selected measurements of urine lead levels in humans can be found in Annex Table AX6-2.11. Urine lead concentrations in the U.S. general population have been monitored in NHANES. Data from the most recent survey (NHANES IV, Centers for Disease Control, 2005) for subjects ≥6 years of age are shown in Table 6-2.4. The geometric mean for the entire sample (n = 2,689) was 0.64 µg/g creatinine (95% CI: 0.60, 0.68). The geometric means for males (n = 1,334) and females (n = 1,335) were 0.64 µg/g creatinine (95% CI: 0.61, 0.67) and 0.64 µg/g creatinine (95% CI: 0.59, 0.69), respectively. These values correspond to approximately 1-1.3 µg lead/day for an adult, assuming a daily creatinine excretion rate of approximately 1.5 g/day in adult females, a body weight of 70 kg for males and 58 kg for females, and a lean body mass fraction of 0.88 for males and 0.85 for females (Forbes and Bruining, 1976; International Commission on Radiological Protection, 1981).

Table 6-2.4. Urine Lead Concentrations in U.S. by Age, NHANES IV (1999–2002)

Age	6–11 years		12–19 years		≥20 years	
Survey Period	1999–2000	2001–2002	1999–2000	2001–2002	1999–2000	2001–2002
n	340	368	719	762	1406	1559
Urine Lead (µg/g)[a]	1.17 (0.98, 1.41)	0.92 (0.84, 1.00)	0.50 (0.46, 0.54)	0.40 (0.38, 0.43)	0.72 (0.68, 0.76)	0.66 (0.62, 0.70) (µg/g)[2]

[a]Urine lead concentrations presented are geometric means (95% CI) of µg lead/g creatinine.

Geometric mean urinary lead excretion rates of 7-10 µg/g creatinine (maximum 43) have been reported in groups of children living in areas impacted by lead smelting operations (Brockhaus et al., 1988). Daily urinary lead excretion can exceed 200 µg/day in association with

occupational exposures (Biagini et al., 1977; Cramer et al., 1974; Lilis et al., 1968; Lin et al., 2001; Wedeen et al., 1975).

6.2.4.4 Urine Lead as a Biomarker of Lead Body Burden

Urine is a major route of excretion of absorbed lead (Chamberlain et al., 1978; Griffin et al., 1975; Kehoe, 1987; Rabinowitz et al., 1976). The kinetics of urinary excretion following a single dose of lead is similar to that of blood (Chamberlain et al., 1978), likely due to the fact that lead in urine derives largely from lead in blood plasma. Evidence for this is the observation that urinary lead excretion is strongly correlated with the rate of glomerular filtration of lead (i.e., glomerular filtration rate × plasma lead concentration; Araki et al., 1986). Estimates of urinary clearance of lead from serum (or plasma) range from 13-22 L/day, with a mean of 18 L/day (Araki et al., 1986; Chamberlain et al., 1978; Manton and Cook, 1984; Manton and Malloy, 1983). Estimates of blood-to-urine clearance, on the other hand, range from 0.03-0.3 L/day with a mean of 0.12 L/day (Araki et al., 1990; Berger et al., 1990; Chamberlain et al., 1978; Gulson et al., 2000; Koster et al., 1989; Manton and Malloy, 1983; Rabinowitz et al., 1976, 1973; Ryu et al., 1983; see Diamond, 1992 for an analysis of these data), consistent with a plasma to blood concentration ratio of approximately 0.005–0.01 L/day (U.S. Environmental Protection Agency, 2003). Based on the above, urinary excretion of lead can be expected to reflect the concentration of lead in plasma and variables that affect delivery of lead from plasma to urine (e.g., glomerular filtration and other transfer processes in the kidney).

Plasma lead makes a small contribution (<1%) to the blood lead concentration and a negligible contribution to total lead body burden. Furthermore, the kinetics of elimination of lead from plasma is fast, relative to lead in bone, where most of the lead burden resides. Therefore, the basic concepts described for blood as a biomarker for body burden also apply to urine. A single urine lead measurement, or a series of measurements taken over short-time span, is likely a relatively poor index of lead body burden (Figure 6-2.8). On the other hand, long-term average measurements of urinary excretion can be expected to be a better index of body burden. In the hypothetical simulation shown in Figure 6-2.8, both the long-term average urinary lead excretion rate and the body burden have approximately doubled.

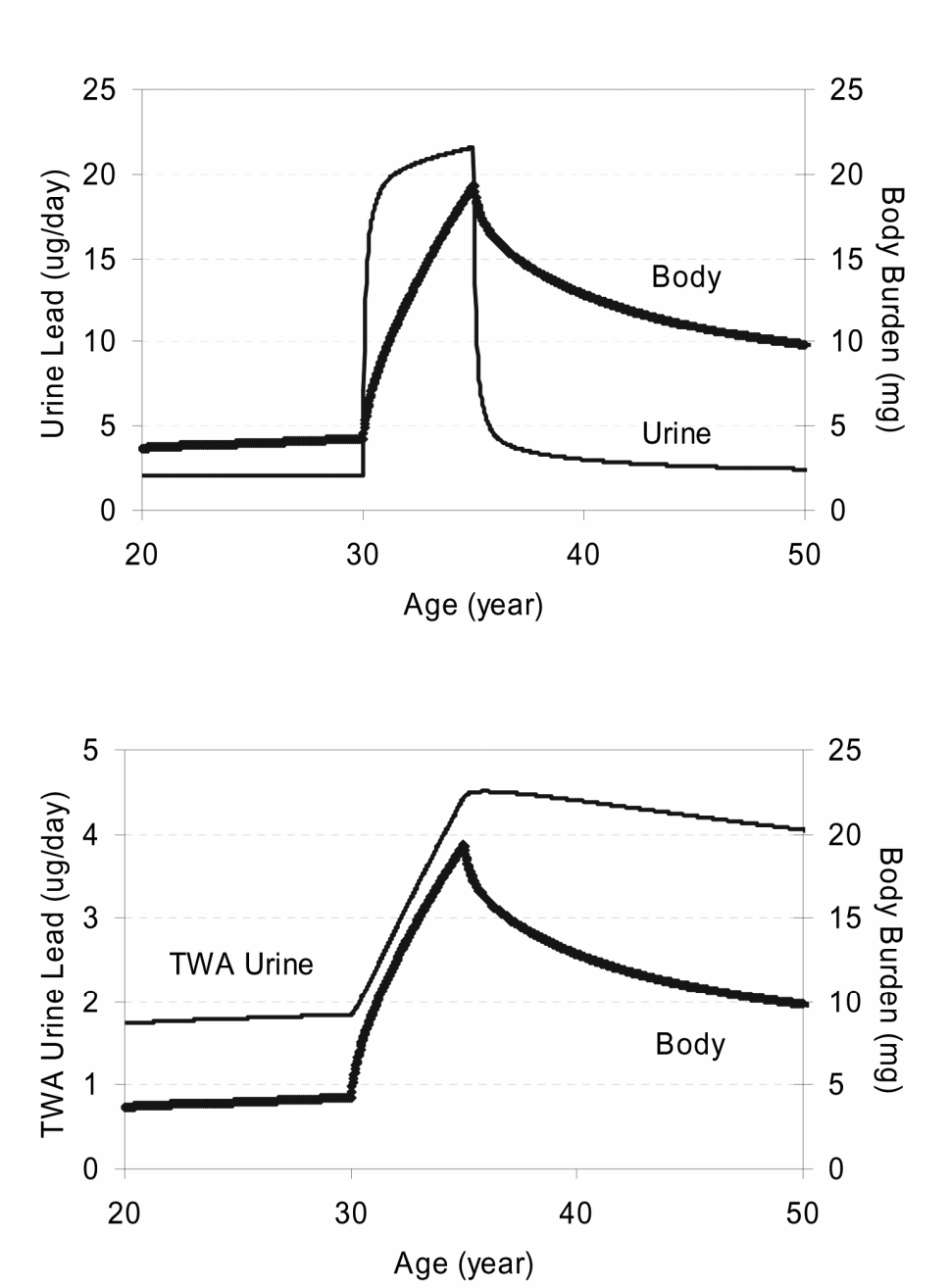

Figure 6-2.8. Simulation of relationship between urinary lead excretion and body burden in adults. An abrupt change in lead uptake gives rise to a relatively rapid change in urinary excretion of lead, to a new quasi-steady state, and a relatively small change in body burden (upper panel). The long-term average urinary lead excretion more closely tracks the pattern of change in body burden (lower panel). Simulation based on Leggett (1993) lead biokinetics model.

6.2.4.5 Urine Lead as a Biomarker of Lead Exposure

Assuming first-order kinetics, a plasma-to-urine clearance (UClP) of 13-22 L/day corresponds to half-time for transfer of lead from plasma to urine of 0.1-0.16 day for a 70 kg adult who has a plasma volume (VP) of approximately 3 L:

$$t_{1/2} = \frac{\mathrm{In}(2) \cdot V_P}{ICl_P}$$

This translates to a very rapid steady-state, much faster than observed for blood lead after a change in exposure level. The kinetics of change in urinary lead excretion in response to a change in exposure, therefore, will be determined by variables that affect the plasma lead level, including partitioning of lead into erythrocytes and exchanges with lead in soft tissues and mobile pools within bone (e.g., bone surface). Here again, the basic concepts that apply to blood lead as a biomarker of exposure also apply to urine lead. Urinary lead excretion reflects, mainly, the exposure history of the previous few months; thus, a single urinary lead measurement cannot distinguish between a long-term low level of exposure or a higher acute exposure. The relationship between urinary lead concentration and lead uptake is thought to be linear, unlike that for blood lead concentration, although there are no direct empirical tests of this assumption in humans. This assumption predicts a linear relationship between lead intake (at constant absorption fraction) and urinary lead excretion rate. Figure 6-2.9 presents a simulated relationship between lead intake and urinary lead excretion in adults and children using both the Leggett (1993) model and O'Flaherty (1993, 1995) model. The major difference between the Leggett model and the O'Flaherty model is in the assignment of the time dependence of bone lead residence. The Leggett model assumes a slow accumulation of a nonexchangable lead pool, whereas the O'Flaherty model assumes a gradual distancing of lead from bone surfaces by diffusion throughout the bone volume (O'Flaherty, 1998).

It is important to emphasize that the above concepts apply to urinary lead excretion rate, not to urinary lead concentration. The concentration of lead in urine (U_{Pb}) is a function of the urinary lead excretion (UE_{Pb}) and the urine flow rate (UFR, L/day):

$$UE_{Pb} = U_{Pb} \cdot UFR$$

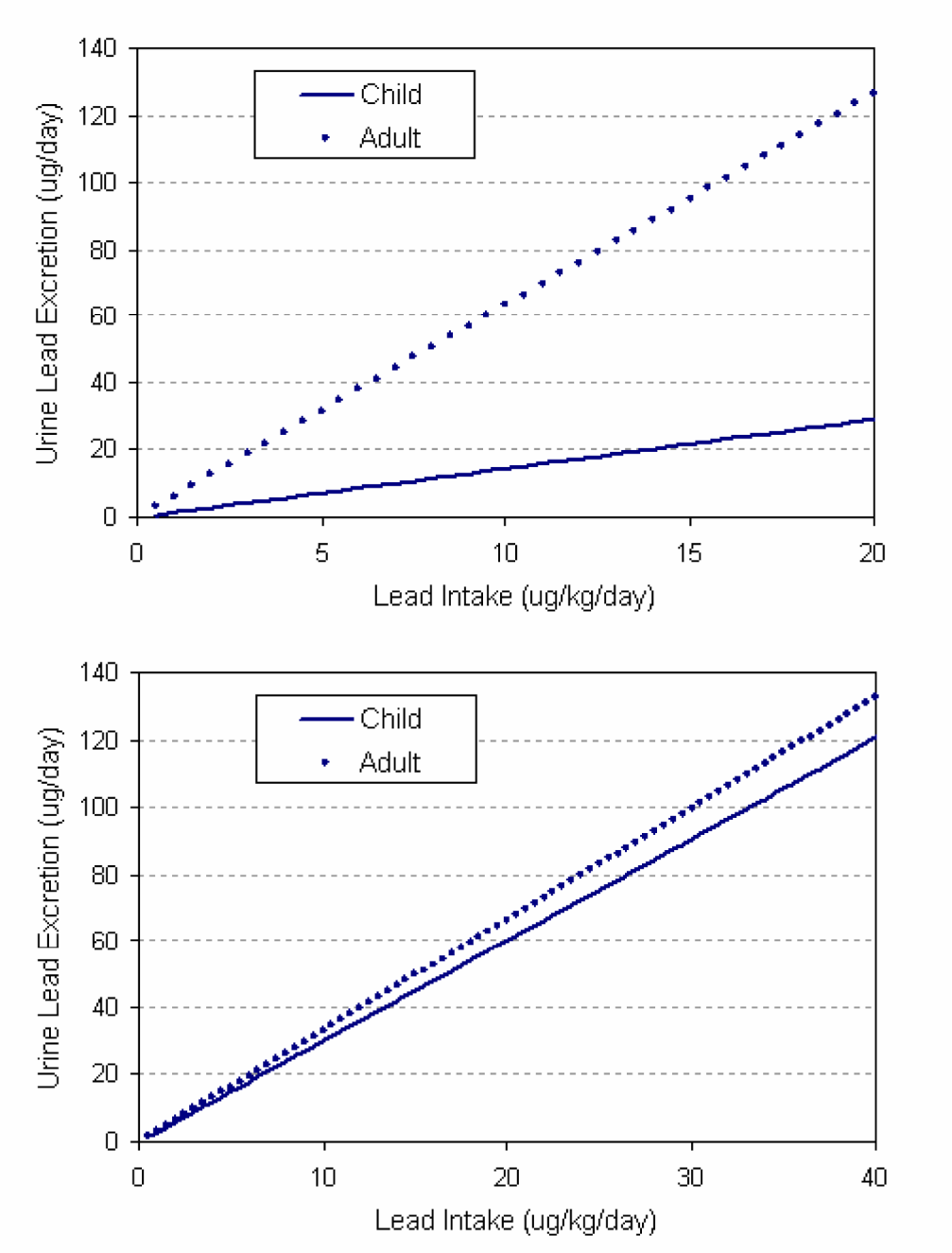

Figure 6-2.9. Simulation of relationship between lead intake and urinary lead excretion in adults and children. Predictions are for a 2-year-old child and 30-year-old adult, for a constant lead intake (µg/kg/day). The relationship is linear, for intake and plasma lead concentration (not shown). Predictions are based on Leggett (1993, upper panel) and O'Flaherty (1993, 1995, lower panel).

Urine flow rate can vary by a factor or more than 10, depending on the state of hydration and other factors that affect glomerular filtration rate and renal tubular reabsorption of the glomerular filtrate. All of these factors can be affected by lead exposure at levels that produce nephrotoxicity (i.e., decreased glomerular filtration rate, impaired renal tubular transport function; see Section 6.4 for discussion of effects of lead on the renal system). Therefore, urine lead concentration measurements provide little reliable information about exposure (or lead body burden), unless they can be adjusted to account for unmeasured variability in urine flow rate (Araki et al., 1990).

A determination of urinary lead excretion rate requires measurement of two variables, urine lead concentration, and urine flow rate; the later requires collection of a timed urine sample, which is often problematic in epidemiologic studies. Collection of un-timed ("spot") urine samples, a common alternative to timed samples, requires adjustment of the lead measurement in urine to account for variation in urine flow (Diamond, 1988). Several approaches to this adjustment have been explored, including adjusting the measured urine lead concentration by the urine creatinine concentration, urine osmolality, or specific gravity (Araki et al., 1990).

The measurement of lead excreted in urine following an injection (intravenous or intramuscular) of the chelating agent calcium disodium EDTA (EDTA provocation) has been used to detect elevated body burden of lead in adults (Biagini et al., 1977; Lilis et al., 1968; Wedeen, 1992; Wedeen et al., 1975) and children (Chisolm et al., 1976; Markowitz and Rosen, 1981). EDTA-provoked urinary lead excretion has been shown to correlate with tibia bone lead measurements (Wedeen, 1992). Given the difficulties associated with the parenteral administration of EDTA, XRF measurements of bone lead, offer a more feasible alternative to the EDTA provocation test for assessment of bone lead stores in epidemiologic studies. More recently, DMSA (DMSA-provocation) has been used as an orally-effective alternative to EDTA and has been applied to epidemiologic studies as dose metric for lead body burden (e.g., Lee et al., 2001; Schwartz et al., 2001a, 2000a, 2000b).

6.2.4.6 Summary of Urine Lead as a Biomarker of Lead Body Burden and Exposure

Similar to blood lead concentration measurements, urinary lead excretion measured in an individual at a single point in time will reflect the recent exposure history of the individual and

physiological variables that determine the plasma lead concentration time profile. Longitudinal measurements of urinary lead excretion can be expected to provide a more reliable measure of exposure history of an individual and will more closely parallel body burden than will single measurements; however, the degree to which this will apply will depend on the sampling frequency with respect to the exposure temporal pattern.

Although, in general, higher urinary lead excretion can be interpreted as indicating higher exposures (or lead uptakes), it does not necessarily predict appreciably higher body burdens. Similar urinary lead excretion rates in two individuals (or populations) do not necessarily translate to similar body burdens or similar exposure histories.

Measurement of the urinary lead excretion rate requires either a timed urine sample, or an approach to adjusting measured urinary lead concentrations for variability in urine flow rate, which by itself may be affected by lead exposure (i.e., lead-induced nephrotoxicity). Both approaches, timed urine samples or adjustment of concentration, introduce complications into the assessment and uncertainties into the interpretation of urinary lead measurements as biomarkers of lead body burden or exposure. The EDTA-provocation test provides a more reliable indicator of elevated body burden than do measurements of basal lead excretion; however, it is not feasible to apply this test for epidemiologic investigations. The DMSA-provocation test may provide a more feasible alternative.

6.2.5 Lead in Hair

6.2.5.1 Summary of Key Findings from the 1986 Lead AQCD

The 1986 Lead AQCD did not discuss applications of hair lead measurements for assessing lead body burden or exposure.

6.2.5.2 Analytical Methods for Measuring Lead in Hair

Methods used for hair lead analysis are summarized in Annex Table AX6-2.1. Wilhelm et al. (1989) reported a detection limit of 0.16 μg/g for GFAAS; use of GFAAS for hair lead measurements has been reported elsewhere (Annesi-Maesano et al., 2003). Gerhardsson et al. (1995a) reported a detection limit of 0.5 μg/g for XRF of the hair shaft; but Campbell and Toribara (2001) found XRF to be unreliable for hair root lead determinations. Use of other

methods has been reported, including ICP (Tuthill, 1996), ET/AAS (Drasch et al., 1997), and AAS (Sharma and Reutergardh, 2000; Esteban et al., 1999).

6.2.5.3 Levels of Lead in Hair

A summary of selected measurements of hair lead levels in humans can be found in Annex Table AX6-2.12. Reported hair lead levels vary considerably. Esteban et al. (1999) reported a geometric mean levels of 5.4 ng/g (range 1-39) for a sample of 189 children (aged 1.9 to 10.6 years) residing in Russian towns impacted by smelter and battery plant operations. By contrast, Tuthill (1996) reported much higher levels in a sample of Boston, MA children (aged 6.5 to 7.5 years, n = 277). Approximately 41% had levels that ranged from 1 to 1.9 µg/g. DiPietro et al. (1989) reported a geometric mean hair lead level of 2.42 µg/g (10–90th percentile range <1.0-10.8) in a general population sample of U.S. adults (aged 20 to 73 years, n = 270). In a post-mortem sample of the general population from Germany (aged 16 to 93 years, n = 150), the median hair lead level was 0.76 µg/g (range 0.026-20.6) (Drasch et al., 1997). Also, Gerhardsson et al. (1995a) reported median values for postmortem samples of 8.0 µg/g (range 1.5-29,000) in active workers (n = 6), 2.6 µg/g (range 0.6-9.3) in retired workers (n = 23), and 2.1 µg/g (range 0.3-96) in a reference group (n = 10).

6.2.5.4 Hair Lead as a Biomarker of Lead Body Burden

Lead is incorporated into human hair and hair roots (Bos et al., 1985; Rabinowitz et al., 1976) and has been explored as a possibly noninvasive approach for estimating lead body burden (Gerhardsson et al., 1995a; Wilhelm et al., 1989, 2002). Hair lead measurements are subject to error from contamination of the surface with environmental lead and contaminants in artificial hair treatments (i.e., dyeing, bleaching, permanents) and are a relatively poor predictor of blood lead concentrations, particularly at low levels (<12 µg/dL) (Campbell and Toribara, 2001; Drasch et al., 1997; Esteban et al., 1999). Studies evaluating quantitative relationships between hair lead and lead body burden have not been reported. Nevertheless, hair lead levels have been used as a dose metric in some epidemiologic studies (e.g., Annesi-Maesano et al., 2003; Esteban et al., 1999; Gerhardsson et al., 1995a; Powell et al., 1995; Sharma and Reutergardh, 2000; Tuthill, 1996).

6.2.5.5 Hair Lead as a Biomarker of Lead Exposure

Rabinowitz et al. (1976) measured hair lead levels in two adult males who received a stable lead isotope supplement to their dietary intake for 124–185 days. Approximately 1% of the daily lead intake was recovered in hair. Temporal relationships between exposure levels and kinetics and hair lead levels, and kinetics of deposition and retention of lead in hair have not been evaluated. Higher hair lead levels were observed in lead workers than in reference subjects with lower blood lead levels (Mortada et al., 2001).

6.2.5.6 Summary of Hair Lead as a Biomarker of Lead Body Burden and Exposure

Although hair lead measurements have been used in some epidemiologic studies, an empirical basis for interpreting hair lead measurements in terms of body burden or exposure has not been firmly established. Hair lead measurements are subject to error from contamination of the surface with environmental lead and contaminants in artificial hair treatments (i.e., dyeing, bleaching, permanents) and, as such, are relatively poor predictor of blood lead concentration, particularly at low levels (<12 μg/dL).

6.3 NEUROTOXIC EFFECTS OF LEAD

This section assesses epidemiologic evidence for neurotoxic effects of lead exposure in children and adults. First presented are studies of the neurotoxic effects of lead on children, with a focus on several prospective studies examining neurocognitive ability. Other topics include measures of academic achievement, cognitive abilities, disturbances in behavior, mood, and social conduct, measures of brain anatomical development and activity, gene-environmental interaction, and reversibility of neurodevelopmental deficits. Then, neurotoxic effects of environmental and occupational lead exposure of adults are discussed.

6.3.1 Summary of Key Findings on Neurotoxic Effects of Lead in Children from 1986 Lead AQCD and Addendum, and 1990 Supplement

The 1986 Lead AQCD stated that children were particularly susceptible to lead-induced neural damage. In particular, human infants and toddlers below that age of 3 years were considered to be at special risk due to their in-utero exposure, increased opportunity for exposure

because of normal mouthing behavior of lead-containing objects, and increased rates of lead absorption due to factors such as iron and calcium deficiencies.

Effective blood lead levels for producing encephalopathy or death in children were noted in the 1986 Lead AQCD as starting at 80–100 μg/dL. Various types of neural dysfunction were stated as being evident at lower blood lead levels. Behavioral (e.g., reaction time, psychomotor performance) and electrophysiological (e.g., altered electrophysiological patterns, evoked potential measures, and peripheral nerve conduction velocities) effects were observed at blood levels as low as 15-30 μg/dL and possibly lower. A concentration-response relationship between blood lead levels and IQ also was observed; a 1-2 point difference in IQ was generally seen with blood lead levels in the 15-30 μg/dL range. However, a study by Schroeder and Hawk (1987) reported a highly significant linear relationship between a measure of IQ and blood lead levels over the range of 6 to 47 μg/dL in a cohort of all African American children of low SES, suggesting that IQ effects might be detected even at these low levels.

The 1986 Addendum discussed the newly published results of several prospective cohort studies on the developmental effects of lead in children. These studies improved upon the previous studies with longitudinal study design that followed children from the prenatal stage, a larger number of subjects, and better analytic techniques to more accurately measure blood lead levels. The four prospective studies (conducted in Boston, MA; Cincinnati, OH; Cleveland, OH; and Port Pirie, Australia) reported significant associations between prenatal and postnatal blood lead levels and neurobehavioral deficits, after adjusting for various potential confounding factors such as maternal IQ and HOME (Home Observation for Measurement of Environment) scores (Bellinger et al., 1984; Dietrich et al., 1986; Ernhart et al., 1985, 1986; McMichael et al., 1986; Vimpani et al., 1985; Wolf et al., 1985). In these studies, the observed maternal and cord blood lead levels were fairly low, with mean levels of approximately 10 μg/dL. These results led the 1986 Addendum to conclude that neurobehavioral deficits, including declines in Bayley Mental Development Index (MDI) scores and other assessments of neurobehavioral function, are associated with prenatal blood lead exposure levels on the order of 10 to 15 μg/dL and possibly even lower, as indexed by maternal or cord blood lead concentrations.

The 1990 Supplement updated evidence from the above-mentioned longitudinal cohort studies and summarized results from other more recent prospective cohort studies conducted in Glasgow, Scotland; Kosovo, Yugoslavia; Mexico City; and Sydney, Australia. Results from

several other international cross-sectional studies also were discussed. The collective evidence from the various prospective cohort and cross-sectional studies reaffirmed the conclusions from the 1986 Addendum that neurobehavioral effects were related to blood lead levels of 10 to 15 µg/dL and possibly lower. Further analyses of the Boston data indicated that deficits in MDI could be detected in relation to cord blood lead levels of 6-7 µg/dL in children within the lower strata for SES (Bellinger et al., 1988). In the Port Pirie study, the relationship between postnatal blood lead levels and MDI at two years of age provided little evidence of a threshold effect (Wigg et al., 1988). Restricting the analysis to children with blood lead levels below 25 µg/dL yielded an even stronger association between integrated postnatal blood lead and McCarthy General Cognitive Index (GCI) scores in the Port Pirie study (McMichael et al., 1988).

Impaired neurobehavioral development was associated with blood lead measures in pregnant women, umbilical cords, and infants up to at least 2 years of age; thus, no distinction could be made as to whether this level of concern applied to only fetuses or infants or preschool-age children. The issue of the persistence of the neurobehavioral effects from low-level lead exposure also was considered. Although the Boston and Cincinnati studies provided limited evidence suggesting that the effects of prenatal lead exposure on neurobehavioral development were not permanent, the evidence available to support this conclusion was inadequate.

6.3.2 Neurotoxic Effects of Lead in Children

Several major developments have occurred in lead research on child neurodevelopment following the 1986 Lead AQCD/Addendum and the 1990 Supplement. First, there has been an attempt to broaden outcome assessments beyond neurocognitive deficits. The earlier emphasis on neurocognitive measures (e.g., MDI, GCI, IQ) in previous studies is understandable from the perspectives of the strong psychometric properties of most of these rigorously standardized measures as well as the immediate pubic health concerns. Examples of other outcomes used to assess neurodevelopment include the number of errors on tests of visual-motor integration, the time required to complete a task assessing manual dexterity, the number of errors and false alarms on a continuous performance test, and the efficiency of short term memory. Additional neurodevelopment outcomes include those which elucidate brain-behavior relationships or the potential real life consequences of early exposure to lead, such as academic and vocational failure and maladjustment to the daily demands of living in a complex society. Thus,

epidemiologic studies of lead neurotoxicity have been expanded to adopt measures of academic achievement, specific cognitive abilities, behavior and mood, sensory acuities, neuromotor function, and direct measures of brain anatomical development and activity. Another development has been the initiation of nutritional and pharmacological intervention studies to assess the impact of treatment on reducing blood lead levels and preventing or moderating the degree of harm to the central nervous systems of young children. Also, in addition to blood and tooth lead, bone lead has emerged as a reliable biomarker of lead exposure. The technology for the assessment of lead in cortical (tibial) and trabecular (patellar) bone using K-shell X-ray fluorescence (XRF) has advanced to the point where it could be applied as a reliable and valid index of cumulative lead dose in neuroepidemiologic studies (Aro et al., 1994).

In recent years, more studies have investigated the impact of blood lead levels below 10 µg/dL on the developing brain. Average blood lead levels in U.S. children ages one to five years decreased from 15 µg/dL to approximately 3 µg/dL between 1976-1980 and 1991-1994, allowing newer studies to examine the effects of low level lead on the neurodevelopment of children (Centers for Disease Control, 2000; Pirkle et al., 1998).

At the time of the last previous criteria review, it was recognized that estimating a threshold for toxic effects of lead on the central nervous system entailed difficulties. As discussed in the 1990 Supplement, insults to the human brain may be irreversible, making it difficult to determine whether any measured insult is the result of current or past exposures. An observed effect concurrent with a measured blood lead concentration may be the result of exposure in the child's earlier life in the womb or infancy. There is also the critical question of reversibility or the persistence of lead effects identified in infants and preschoolers into school age and later. A given effect observed at younger ages may not persist due to functional compensation or a return to a normal neuromaturational trajectory (Dietrich et al., 1990). Another problem is that it is sometimes difficult to distinguish between neurobehavioral effects due to lead and effects owing to the many social, economic, urban-ecological, nutritional, and other medical factors that are known to have important effects on neurobehavioral development. Equally important is the high probability that the concentration-response relationship and even the neurobehavioral lesion associated with childhood lead exposure may vary as a function of these cofactors (Bellinger, 1995).

In the following sections, prospective cohort studies and cross-sectional studies of neurocognitive ability published since the 1990 Supplement are presented first. Then, studies examining the effect of lead on a variety of neurodevelopmental outcomes, including academic achievement; specific cognitive abilities; disturbances in behavior, mood, and social conduct; sensory acuities; neuromotor function; and brain anatomical development and acuity, are discussed. This is followed by a presentation of issues involved in understanding lead neurotoxicity in children, including gene-environment interactions, reversibility of lead effects, times of vulnerability, and potential threshold levels for effects.

6.3.2.1 Neurocognitive Ability

6.3.2.1.1 Prospective Longitudinal Cohort Studies of Neurocognitive Ability

Several prospective longitudinal cohort studies were initiated in the 1980s because it became widely recognized that the cross-sectional study design was inadequate to address a number of research issues (U.S. Environmental Protection Agency, 1986; World Health Organization, 1977). These longitudinal studies were characterized by serial measures of dose (blood lead levels) spanning (in most cases) the prenatal and postnatal periods of central nervous system development, thus helping to clarify the temporal association between exposure and insult. Also, developmental assessments that extended into the school-age period were planned to determine if early lead associated neurobehavioral impairments were permanent or reversible in the fullness of time. It was also determined that assessment of potential confounding factors should be comprehensive and include measures of perinatal health, nutrition, maternal consumption of other neurotoxicants during pregnancy, parental intelligence, and direct observations of parenting behavior. These studies were also characterized by very careful attention to biostatistical issues and strategies (Bellinger, 1995; Ernhart, 1995).

At the time of the 1990 Supplement, studies were underway or planned in the U.S., Australia, Scotland, the former Yugoslavia, and Mexico. These cohorts differed in the source and degree of lead exposure and in other important aspects, notably ethnicity and SES. Nevertheless, the early results from several of these studies have been largely responsible for the emergence of the current perspective that blood lead concentrations as low as 10 μg/dL, or perhaps even lower, may pose a risk for neurodevelopmental toxicity (Davis and Svendsgaard, 1987; U.S. Environmental Protection Agency, 1990). Most of the prospective studies underway

in 1990 continued to follow their subjects into the later preschool and school age years with age-appropriate measures of intelligence. Continued follow-up of these cohorts was important due to the following: (1) greater reliability and precision of measurements attained with assessments of older children; (2) high predictability of adult intellectual functioning from measures of IQ in the older child; and (3) examination of potential effects of lead on important abilities that cannot be easily tapped during infancy such as executive functions and higher order reasoning (McCall, 1979).

A unique aspect of this research was that most investigators agreed during the formative stages of their projects to develop somewhat similar assessment protocols (Bornschein and Rabinowitz, 1985). This has facilitated comparison of results across studies and allowed for sophisticated meta- and pooled-analyses of these data (e.g., Pocock et al., 1994; Schwartz, 1994; World Health Organization, 1995; Lanphear et al., 2005; Rothenberg and Rothenberg, 2005).

In the following sections, further updates on the individual prospective cohort studies are presented in chronological order of study initiation. The prospective cohort studies reviewed are summarized in Annex Table AX6-3.1. Results of the meta- and pooled-analyses are presented later in this section.

Boston Study

In the 1986 Addendum, the most advanced investigation at that time was the Boston Prospective Study (Bellinger et al., 1984). The subjects were 216 middle-to upper-middle-class Boston children, 90% of whom had cord blood lead levels below 16 µg/dL (maximum 25 µg/dL). Cord blood lead levels in the "high" group (mean 14.6 µg/dL) were associated with lower covariate-adjusted scores on the Mental Development Index (MDI) of the Bayley Scales of Infant Development at 6 months of age. It was concluded that although lower level lead exposure in utero may result in delays in early sensorimotor development, the Boston results did not allow estimation of the persistence of these effects nor the public health significance of the findings. The association between higher cord blood lead and lower MDI persisted to 24 months; however no association was observed between postnatal blood lead levels and MDI (Bellinger et al., 1985, 1986).

Particular attention was focused on the Boston study, which was among the more mature in terms of follow-up, in the 1990 Supplement (Bellinger et al., 1987; Bellinger et al., 1991).

With respect to the effects of cord blood lead concentrations on MDI assessed longitudinally from 6 to 24 months, the lead associated deficits were evident across the entire range of blood lead levels starting at 10 μg/dL, which reinforced the previous designation of 10-15 μg/dL as a blood level of concern for early neurodevelopmental deficits. At approximately 5 years of age, cord blood lead levels were not significantly associated with the McCarthy GCI, but blood lead level at 2 years of age (mean 6.8 μg/dL [SD 6.3]) was significantly associated with lower scores. Although cord blood lead concentrations were not independently associated with deficits in 5-year neurocognitive status, the risk of obtaining lower GCI scores was greater among subjects with higher prenatal and postnatal blood lead concentrations. Boston investigators also examined the relationship between lead measured in shed deciduous teeth obtained from 102 children in their cohort (mean 2.8 ppm [SD 1.7]) and GCI at 5 years of age. Prior to covariate-adjustment, there was a very strong and significant relationship amounting to a loss of more than 10 points in GCI for each log increment in dentine lead. However, in the multivariable analysis the tooth lead coefficient, although negative, was no longer statistically significant. Reduced sample size should be taken into consideration in interpreting this null finding.

Since the 1990 Supplement, the Boston investigators reexamined 148 of their subjects at 10 years of age with the Wechsler Intelligence Scale for Children-Revised (WISC-R) and other neurobehavioral assessments (Bellinger et al., 1992). They examined the association of WISC-R scores at 10 years of age with blood lead concentrations in the cord blood and at 6 months, 12 months, 18 months, 24 months, 57 months, and 10 years. Only blood lead levels at 24 months were significantly associated with full scale and verbal IQ and marginally associated with performance IQ, after adjusting for HOME score, maternal age, birth weight, and maternal IQ. The integrated average blood lead level in this cohort over the first 2 years was 7.0 μg/dL (range 4-14 μg/dL). An increase of 10 μg/dL in blood lead level at age 2 was associated with a decrement of 5.8 points (95% CI: 1.8, 9.9) in full scale IQ. These findings indicated that children's performance was much more strongly associated with blood lead levels at age 2 than with blood lead levels at other ages. It is unclear whether this reflects a special vulnerability of the nervous system during this period or simply the fact that blood lead level tends to peak in the second year.

A reanalysis involving the total Boston cohort that employed nonparametric smoothing revealed that the inverse association persisted at blood lead levels below 5 µg/dL (Schwartz, 1994). Bellinger and Needleman (2003) reanalyzed data on 48 children whose measured blood lead concentrations never exceeded 10 µg/dL. Reduction in full scale IQ at 10 years was significantly associated with blood lead levels at 2 years of age following covariate adjustment. A larger deficit of 15.6 points (95% CI not presented) per 10 µg/dL increase in blood lead levels was observed in this cohort, compared to the 5.8 point deficit observed in the entire cohort. These findings indicated that the inverse slope might be steeper at blood lead levels below 10 µg/dL.

Cincinnati Study

Interim results on a partial sample of 185 subjects from a cohort of 305 were available from the Cincinnati prospective study in the 1986 Addendum and the 1990 Supplement (Dietrich et al., 1986, 1987a). The Cincinnati study investigators reported an inverse relationship between prenatal maternal blood lead levels (mean 8.3 µg/dL) and 6 month Bayley MDI. This effect was mediated, in part, through lead-associated reductions in birth weight and gestational maturity. A more complete analysis of the full Cincinnati cohort confirmed these interim findings (Dietrich et al., 1987b).

Further updates of the Cincinnati study appeared after the 1990 Supplement. The Kaufman Assessment Battery for Children (KABC) was administered to approximately 260 children at 4 and 5 years of age (Dietrich et al., 1991; 1992). The principal findings at 4 years were that higher neonatal blood lead concentrations were associated with poorer performance on all KABC subscales. However, this relationship was confined to children from the poorer families. Following full covariate adjustment, few statistically significant relationships remained. At 5 years of age, postnatal blood lead levels were associated with performance on all subscales of the KABC; however, few statistically significant relationships remained after adjustment for covariates. Nevertheless, it is of interest that at both 4 and 5 years the KABC subscale that assessed visual-spatial skills was among those that remained the most highly associated with various indices of postnatal exposure following covariate adjustment. At the age of approximately 7 years, 253 children in the Cincinnati cohort were administered the WISC-R (Dietrich et al., 1993a). In this cohort, approximately 35% had at least one blood lead

concentration ≥25 µg/dL while 95% exceeded 10 µg/dL sometime during the first 5 years of life. Postnatal blood lead concentrations were inversely associated with full scale and performance IQ, after adjusting for HOME score, maternal IQ, birth weight, birth length, child gender, and cigarette consumption during pregnancy. Figure 6-3.1 presents the unadjusted and adjusted concentration-response relationship between lifetime average blood lead concentrations and performance IQ. Following covariate adjustment, a statistically significant relationship was observed between postnatal blood lead levels at 5 and 6 years of age and full scale IQ. Postnatal blood lead levels at nearly all ages (including the integrated average blood lead level) were inversely associated with performance IQ. Blood lead levels at 6 years of age were most strongly associated with performance IQ – a 5.2 point [95% CI: 2.3, 8.1] decline was observed for each 10 µg/dL increase in blood lead level. A 10 µg/dL increase in lifetime average blood lead concentration was associated with a 2.6 point (95% CI: 0.2, 5.0) decline in performance IQ.

At 15-17 years of age, the Cincinnati subjects were administered a comprehensive neuropsychological battery (Ris et al., 2004). Variables derived from the Cincinnati neuropsychological battery were subjected to a principal components factor analysis that yielded five factors, including a learning/IQ factor that had high loadings for the Vocabulary and Block Design subtests from the WISC-III as well as the Reading, Spelling, and Arithmetic subscales of the Wide Range Achievement Test-Revised (WRAT-R). Prenatal, Average Childhood, and 78 month blood lead levels were used in a series of multiple regression analyses. Following covariate-adjustment, there was a trend towards significance for higher blood lead concentrations in later childhood (e.g., 78 months) to be associated with lower learning/IQ factor scores, but this was largely observed in subjects from the lower end of the socioeconomic scale in the sample. This finding is consistent with previous reports that children in the lower social strata may be more vulnerable to general effects on cognitive development and learning (Bellinger, 2000; Winneke and Kraemer, 1984).

Cleveland Study

Early results of the Cleveland prospective study also were reviewed in the 1986 Addendum and 1990 Supplement. By selection, about half of the mothers had histories of alcohol abuse as measured by the Michigan Alcoholism Screening Test. The other women were matched controls. The initial cohort included 389 infants with a mean cord blood lead level of

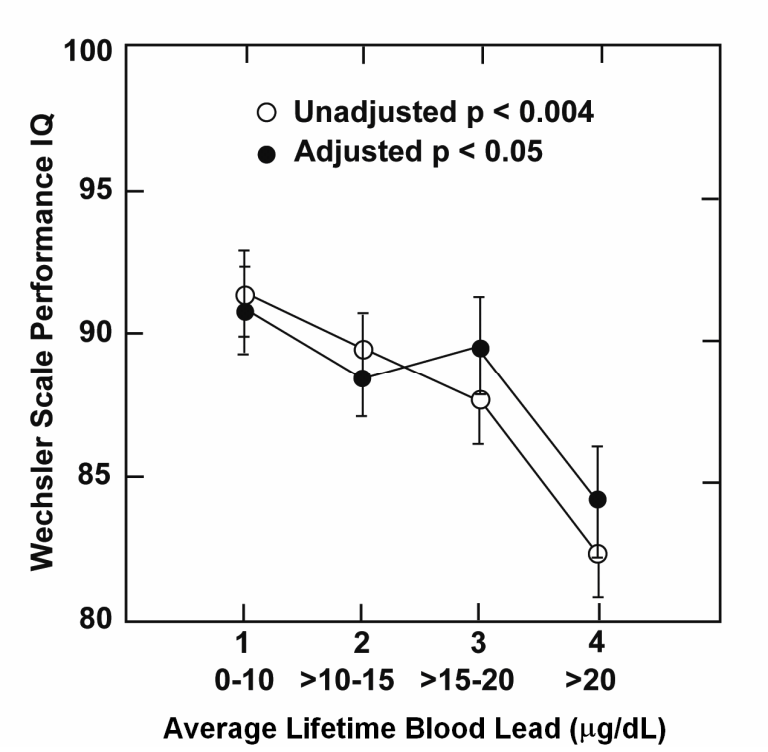

Figure 6-3.1. Unadjusted and adjusted relationships between average lifetime blood lead concentrations and Wechsler Scale performance IQ. Mean ± SD lifetime average blood lead concentrations within each category were as follows: 0-10 µg/dL, 7.7 ± 1.4 µg/dL (n = 68); >10-15 µg/dL, 12.3 ± 1.4 µg/dL (n = 89); >15-20 µg/dL, 17.1 ± 1.2 µg/dL (n = 53); and >20 µg/dL, 26.3 ± 5.0 µg/dL (n = 41).

Source: Dietrich et al. (1993a).

5.84 µg/dL (maximum 14.7). In addition to size, minor morphological anomalies, and 1- and 5-minute Apgar performances, infants were evaluated on the Brazelton Neonatal Behavioral Assessment Scale (NBAS) and part of the Graham-Rosenblith Behavioral Examination for Newborns (G-R). Of the 17 neonatal outcomes examined, the NBAS Abnormal Reflexes scale and neurological soft signs assessed by G-R were associated with cord blood lead levels in the range of 3 to 15 µg/dL following covariate adjustment (Ernhart et al., 1986). A follow-up study observed a significant effect of the neurological soft signs measure on Bayley MDI scores at 12 months; however, prenatal lead exposure was not associated with MDI scores at 6-24 months or Stanford-Binet IQ (S-B IQ) scores at 36 months (Wolf et al., 1985).

In 285 children from the original cohort, maternal and cord blood lead levels, as well as postnatal blood lead levels at 6 months, 2 years, and 3 years were examined in relation to Bayley MDI, Psychomotor Index (PDI), and Kent Infant Development Scale (KID) at 6 months, MDI at 1 year and 2 years, and S-B IQ at 3 years of age (Ernhart et al., 1987, 1988). After covariate adjustment, only maternal blood lead level at delivery (mean 6.5 µg/dL [maximum 11.8]) was inversely associated with MDI, PDI, and KID scores at 6 months. No other indices of prenatal or postnatal lead exposure were inversely associated with assessments of global intellectual functioning. Language development also was assessed in the Cleveland cohort at 1, 2, and 3 years of age. Few significant associations remained after covariate adjustment (Morrow-Tlulak and Ernhart, 1987).

At 4 years and 10 months, 242 children from the Cleveland cohort were administered the Wechsler Preschool and Primary Scale of Intelligence (WPPSI) test. Significant negative correlations were observed between full scale, verbal, and performance IQ, and prenatal (both maternal and cord) and postnatal (at 2 years and 3 years) blood lead levels (Ernhart and Morrow-Tlulak, 1987). However, these associations were no longer significant after adjustment for various covariates, including HOME score, maternal IQ, parent education, race, medical problems, maternal alcohol use in pregnancy, Michigan Alcoholism Screening Test score, maternal use of marijuana, and several categories of psychosocial trauma scale. The authors reported a large contribution of these covariates to the variance of the WPPSI scores; the smaller effects of lead may have been suppressed by these other social factors. In particular, the HOME score was found to most strongly contribute to the child's IQ. In 164 children, shed deciduous incisor were collected between ages 5 and 7. Circumpulpal dentine lead levels were found to be significantly associated with full scale, verbal and performance IQ, assessed using the WPPSI test, at 4 years and 10 months, after adjustment for various covariates except for HOME score. After additional adjusting for HOME score, the lead effects on all three IQ measures diminished, but remained statistically significant for verbal IQ ($p = 0.01$) and marginally significant for full scale IQ ($p = 0.06$). An increase in dentine lead from the 10th percentile to the 90th percentile level (13.5 µg/g to 129.4 µg/g) was associated with a 6.0 point (95% CI: 1.4, 10.6) decrease in verbal IQ and a 4.5 point (95% CI: −0.2, 9.2) decrease in full scale IQ. The estimated lead effect increased as a function of the level of measurement error in the dentine lead variable. This finding of an adverse effect for dentine lead is not consistent with previous analyses of the

Cleveland study showing that blood lead levels are, generally, not associated with cognitive outcomes after covariate adjustment.

Port Pirie, Australia Study

Preliminary results from the Port Pirie, Australia study also were described in the 1986 Addendum (Vimpani et al., 1985). Lower Bayley MDI scores at 2 years from 592 children were significantly associated with higher integrated postnatal blood lead levels (approximately 20% of the sample had blood lead levels >30 μg/dL at the time of assessment), but not with maternal prenatal, delivery, or cord blood lead levels. Results of this interim analysis were interpreted with caution since important covariates such as maternal IQ and HOME scores were not available for the entire cohort at the time of the analyses.

The Port Pirie cohort study had reported results out to 4 years when the 1990 Supplement was released (McMichael et al., 1988). Following adjustment for covariates, lead concentrations at most postnatal sampling points as well as an integrated average for the 4-year postnatal period were significantly and inversely associated with scores on the McCarthy Scales of Children's Abilities. The GCI scores declined by approximately 4.5 points (95% CI: 0.2, 8.8) for a doubling in blood lead levels. Similar deficits occurred in the perceptual-performance and memory scores. The integrated postnatal blood lead levels among the 537 children in this cohort were among the highest of the prospective studies (geometric mean 19 μg/dL). However, further analyses indicated that the effects observed did not depend on children with the more extreme levels of exposure. The concentration-response relationship between blood lead and GCI was stronger among children with blood lead levels below 25 μg/dL than it was overall.

Of all of the prospective studies of lead and child development, the Port Pirie cohort study was probably among the best positioned to reliably detect effects of low level lead exposure into later childhood owing to its wide range of exposure, large sample size, and lack of extremes in terms of sample social advantage or disadvantage. The WISC-R was administered to 494 children between 7 and 8 years of age (Baghurst et al., 1992). IQ scores were examined in relation to ln-transformed blood lead concentration. Following adjustment for covariates there was little association with pre- and perinatal lead exposure assessments. However, significant decrements in full scale and verbal IQ were found to be associated with postnatal blood lead levels. The estimated effect size was a loss of 3.3 points (95% CI: 0.2, 6.5) in full scale IQ and

4.0 points (95% CI: 0.7, 7.2) in verbal IQ in association with a doubling of the integrated postnatal blood lead concentration up to three years. In light of the Cincinnati findings, it is of interest that the Block Design subtest of the WISC-R (a measure of visual-spatial abilities), exhibited the strongest association with lead exposure. Port Pirie investigators also collected deciduous central upper incisors from 262 children in their cohort (McMichael et al., 1994). After covariate adjustment, a significant inverse association was observed between tooth lead concentration and WISC-R full scale IQ at 7 years of age. The adjusted estimated decline in full scale IQ across the tooth lead range from 3 to 22 μg/g (range for 90% of population) was 5.1 points (90% CI: 0.2, 10.0). Once again, the Block Design subtest was among the most highly sensitive.

Port Pirie children were assessed again at 11-13 years of age to examine the persistent relationship between exposure to environmental lead and intelligence (Tong et al., 1996). At that age, Port Pirie investigators were able to recall 375 children for IQ assessments. At 11-13 years of age, the geometric mean lifetime average blood lead concentration was 14.1 μg/dL. WISC-R scores were significantly and inversely associated with integrated lifetime average blood lead concentrations out to 11-13 years. Later blood lead concentrations after 3 years of age were more predictive of lower IQ. Mean full scale IQ declined by 3.0 points (95% CI: 0.1, 5.9) for a doubling of lifetime average blood lead concentrations. The authors could find no clear evidence of a threshold level in their data.

Sydney, Australia Study

Unlike Port Pirie, the reports on the Sydney cohort study were consistently negative with respect to the effects of exposure on neurodevelopment (Cooney et al., 1989a,b; McBride et al., 1989). In the 298 mothers and infants sampled, geometric mean blood lead levels at delivery were 9.1 μg/dL and 8.1 μg/dL, respectively, with less than 2% in excess of 15 μg/dL. Mean postnatal blood lead levels peaked at 16.4 μg/dL when children reached 18 months and then declined to 10.1 μg/dL at 48 months. No significant, inverse relationships were reported between prenatal or postnatal blood lead concentrations and neurodevelopmental assessments conducted from 6 months through 4 years of age. The McCarthy Scales of Children's Abilities was administered to 207 children at 4 years of age but no associations with blood lead levels were observed prior to or following covariate-adjustment. As in the case of the Cleveland study,

the authors noted that the HOME score was a strong contributor to the neurodevelopmental assessments at all ages. As stated in the 1990 Supplement, this raises the questions of whether lead exposure might have covaried with HOME scores. If so, adjusting for HOME scores would reduce the statistical power to examine the effect of postnatal blood lead levels on the neurocognitive measures. Also note that the interpretation of the Sydney findings has been complicated by concerns about possible contamination of capillary blood lead samples collected during the early phases of the investigation (Cooney et al., 1989b).

The Sydney prospective study further assessed 175 subjects that remained in the study at 7 years of age (Cooney et al., 1991). Geometric mean blood lead concentrations peaked at 2 years of age (15.2 µg/dL). The geometric mean blood lead level at 7 years of age was 7.7 µg/dL. The WISC-R and other neurobehavioral assessments were administered. The adjusted correlations between postnatal blood lead levels and WISC-R scores were consistently negative but nonsignificant at the $p = 0.05$ level. The r value (units = SD of IQ per SD of blood lead) for the correlation between full scale IQ and concurrent blood lead at age 7 years was −0.06 (95% CI: −0.20, 0.09). The correlation coefficient is not significantly different form Bellinger et al. (1992) for 57-month-old children, −0.07 (95% CI: −0.23, 0.08), or from Lanphear et al. (2005) for children aged 4.8 to 10 years, −0.20 (95% CI: −0.28, −0.12). All correlation coefficients are for full scale IQ and concurrent blood lead concentrations.

Results from this follow-up study were consistent with their earlier reports of no association between blood lead levels <15 µg/dL and developmental deficits. However, the authors noted that their study was not designed to examine small deficits associated with blood lead levels at this magnitude. They reported that the size of their cohort did not provide sufficient power to detect effects less than 5%. Cooney et al. concluded that results from their study indicate that if developmental deficits do occur at blood lead levels below 25 µg/dL, the effect size is likely to be less than 5%.

Mexico City Study

Preliminary results of the Mexico City cohort prospective study were presented in the 1990 Supplement (Rothenberg et al., 1989). Blood lead levels from 42 mother-infant pairs were measured at 36 weeks of pregnancy (mean 15.0 µg/dL) and delivery (mean 15.4 µg/dL), and in the cord blood (mean 13.8 µg/dL). The Brazelton NBAS was administered to infants at

48 hours, 15 days, and 30 days after birth. None of the lead measures were associated with the NBAS outcomes; however, several differential lead measures (i.e., maternal blood lead at 36 weeks of pregnancy minus cord blood lead) were found to be associated with several outcome variables. Increases in the blood lead of the mother during the last month of pregnancy or a cord blood lead level higher than the mother's blood lead level were associated with adverse changes in Regulation of States, Autonomic Regulation, and Gestation Age.

Schnaas et al. (2000) further examined the effect of postnatal blood lead level on cognitive development in 112 children with complete data from the Mexico City study. Lead was measured in blood every 6 months from 6 to 54 months. Intellectual status was assessed with the McCarthy GCI. The purpose of the study was to estimate the magnitude of the effect of postnatal blood lead level on the GCI and describe how the effect varies with the time between blood lead measurements and the neurocognitive assessments. The geometric mean blood lead level between 24-36 months was 9.7 µg/dL (range 3.0-42.7). A number of significant interactions were observed between blood lead levels and age of assessment. The greatest effect was found at 48 months, with a decrease of 4.0 points (95% CI not presented) in adjusted GCI score being observed for a doubling of the 24-36 month blood lead level. The authors concluded that 4 to 5 years of age (when children are entering school) appears to be a critical period for the manifestation of earlier postnatal blood lead level effects.

In a related study, Gomaa et al. (2002) examined prenatal and postnatal lead exposure effects on the neurodevelopment of 197 children aged 2 years residing in Mexico City. Lead was measured in the umbilical cord and maternal venous blood samples at delivery. Maternal body burden was measured by obtaining cortical (tibial) and trabecular (patellar) bone lead measurements using K-shell XRF within 4 weeks of delivery. At 2 years of age, the Bayley MDI and PDI were administered. The major objective of this study was to compare lead levels in umbilical cord blood and maternal bone as independent predictors of infant mental development. Mean blood lead concentrations in the cord blood, at 12 months of age, and at 24 months at age were 6.7 µg/dL (SD 3.4), 7.2 µg/dL (SD 2.8), and 8.4 µg/dL (SD 4.6), respectively. Mean maternal patella and tibia bone lead levels were 17.8 µg/g (range <1-76.6) and 11.5 µg/g (range <1-85.9), respectively. Following covariate adjustment, postnatal blood lead concentrations were not significantly associated with MDI; however, lead levels in cord blood and trabecular bone were found to be significantly associated with lower scores on the

Bayley MDI. Maternal trabecular bone lead levels predicted poorer sensorimotor functioning in children 2 years of age independent of the cord blood lead level. The authors concluded that higher maternal trabecular bone lead concentrations constitute an independent risk factor for impaired mental development in infants at 2 years of age and that this is likely due to the mobilization of maternal bone lead stores over the course of gestation.

Kosovo, Yugoslavia Study

The neurodevelopment results of a large birth cohort study of 577 children in two towns in Kosovo, Yugoslavia were not available at the time of the 1990 Supplement. The study took place in Titova Mitrovica, near the site of a longstanding lead smelter, refinery, and battery plant, and in Pristina, a less exposed community 25 miles to the south. A unique characteristic of this cohort was the high prevalence of anemia secondary to iron deficiency (34% with hemoglobin concentrations <10.5 μg/dL at 2 years of age). The investigators began providing iron-fortified multivitamin supplements to the entire cohort when the children were between 18 to 38 months of age (Wasserman et al., 1994).

Like Port Pirie, this was one of the more highly exposed cohorts. Blood lead levels were obtained during the second trimester, at delivery, from the umbilical cord and postnatally at 6-month intervals to 90 months. At birth, geometric mean cord blood lead levels were nearly 21 μg/dL in the smelter area (Wasserman et al., 1992). At age 2 years, geometric mean blood lead concentrations were 35.5 μg/dL and 8.4 μg/dL among infants from Titova Mitrovica and Pristina, respectively.

Neurocognitive measures of mental abilities were administered at 2, 4, 7, and 10-13 years of age. The relationships between these neurocognitive outcomes and log-transformed blood lead levels were assessed. A doubling of blood lead levels at 2 years of age was associated with a covariate-adjusted decline of 1.6 points (95% CI: 0.2, 3.0) in Bayley MDI. Statistically nonsignificant decrements in MDI were associated with blood lead levels measured at all other time points. Iron deficiency anemia also was an independent predictor of lower MDI (Wasserman et al., 1992). When examined at 4 years of age, the geometric mean blood lead concentration of children from the smelter area was 39.9 μg/dL, while the geometric mean for children in the "unexposed" area was 9.6 μg/dL (Wasserman et al., 1994). Children were administered the McCarthy Scales of Children's Abilities. Higher prenatal and cord blood lead

concentrations were associated with lower GCI scores. Following covariate-adjustment, children of mothers with prenatal blood lead levels greater than 20 μg/dL scored a full standard deviation below children in the lowest exposure group (<5 μg/dL prenatal blood lead). A statistically significant association also was observed between nearly every blood lead measurement (at 6-month intervals since birth) and GCI. At 4 years of age, a doubling of blood lead levels was associated with a reduction of 2.8 points (95% CI: 1.4, 4.3) on the GCI. The Perceptual-Performance subscale of the McCarthy was found to be most sensitive to lead exposure.

When 301 children were examined at 7 years of age with the WISC-III, significant associations were observed between postnatal blood lead concentrations and IQ, with consistently stronger associations between performance IQ and later blood lead measures (Factor-Litvak, 1999). The adjusted intellectual loss associated with a doubling in lifetime average blood lead was 2.7 points (95% CI: 1.7, 3.7) in full scale IQ, 2.8 points (95% CI: 1.7, 4.0) in performance IQ, and 2.1 points (95% CI: 1.1, 3.2) in verbal IQ. By 7 years, measures of iron status were no longer significantly associated with IQ.

At age 10-12 years, 290 subjects with complete data on exposure and covariate factors were assessed again with the WISC-III (Wasserman et al., 2003). However, in addition to well-characterized exposure histories based on serial blood lead assessments, tibial bone lead was measured using ^{109}Cd based K-shell XRF (Todd et al., 2001) on a representative subsample of 167 subjects from both communities. Blood lead and bone lead measures were highly correlated in Titova Mitrovica, but not in Pristina. Following covariate-adjustment, average lifetime blood lead level was significantly and negatively related to all components of WISC-III IQ. A doubling of average blood lead concentration was associated with a decrease in full scale, performance, and verbal IQ of 1.6 points (95% CI: 0.4, 2.8), 1.5 points (95% CI: 0.3, 2.8), and 1.5 points (95% CI: 0.3, 2.6), respectively. The relationships between bone lead and IQ scores were stronger than those for blood lead, at least in the more highly exposed smelter community. For each doubling of tibial bone lead concentrations, full scale, performance, and verbal IQ decreased by an estimated 5.5, 6.2, and 4.1 points, respectively. The authors also reported that significant associations between tibial lead concentrations and IQ scores persisted despite inclusion of blood lead into the model. The inference drawn from these findings was that associations between bone lead and IQ outcomes may be stronger than those between blood lead measures and IQ.

Shanghai, China Study

A prospective study of low-level prenatal and postnatal exposure was initiated in 1993 by Shen et al. (1998) in Shanghia, China. Pregnant women were recruited from a maternal and child health care facility in the community. Lead levels were determined on 348 cord blood samples. The geometric mean cord blood lead level was 9.2 µg/dL (range 1.6-17.5); 40.8% of the infants had cord blood lead levels ≥10 µg/dL. Infants were further selected for study on the basis of their cord blood lead concentrations – the low lead group (n = 64) had levels <30th percentile while the high lead group (n = 69) had levels >70th percentile. Mean cord blood lead concentrations in the high lead group and low lead group were 13.4 µg/dL (SD 2.0) and 5.3 µg/dL (SD 1.4), respectively. At 3, 6, and 12 months, infants were administered the Chinese version of the Bayley Scales of Infant Development. Capillary blood samples were collected at each visit to ascertain levels of postnatal exposure. Mean blood lead at 1 year of age was 14.9 µg/dL (SD 8.7) in the high lead group and 14.4 µg/dL (SD 7.7) in the low lead group. Postnatal blood lead levels were not significantly different in the high and low lead groups.

At all three ages, the Bayley MDI, but not PDI, was associated with cord blood lead groupings following adjustment for covariates, which included a wide range of perinatal, demographic, social, and environmental factors. Postnatal blood lead concentrations were not associated to any Bayley measures. Differences in mean MDI between cord blood lead groups were 3.4 points at 3 months ($p = 0.02$), 6.3 points at 6 months ($p = 0.03$), and 5.2 points at 12 months ($p = 0.03$). The early results of this prospective study are generally in accord with similar investigations in Boston, Cincinnati, and Cleveland. The authors concluded that the adverse effects of prenatal lead exposure on early neurobehavioral development are readily discernible and stable over the first year of life.

Rochester Study

The Rochester prospective study, initiated in 1994, examined the relationship between blood lead levels and IQ at 3 and 5 years of age in 172, predominantly African-American, lower SES children (Canfield et al., 2003a). Participants were enrolled when children were 5 to 7 months of age in what was originally a study of lead dust control methods (Lanphear et al., 1999). Blood lead concentrations were assessed at 6-month intervals until 2 years and annually thereafter. No data were available on prenatal exposure. The measure of IQ was the abbreviated

Stanford-Binet Intelligence Scale-4th Edition (SBIS-4). Potential confounders assessed included gender, birth weight, iron status, HOME scores, maternal IQ, SES, and tobacco use during pregnancy.

Blood lead concentrations in the Rochester cohort were quite low for an urban population as this study was conducted after public health measures to reduce blood lead levels in children were already having a dramatic impact in the U.S. population. Blood lead levels peaked at 2 years of age (mean 9.7 μg/dL). The mean lifetime average blood lead concentration was 7.7 μg/dL at the age of 3 years and 7.4 μg/dL at the age of 5 years. At 5 years of age, 56% of the children had a peak blood lead concentration below 10 μg/dL. Following adjustment for covariates, there were significant inverse associations with full scale IQ at both 3 and 5 years of age for all blood lead variables, including lifetime average up to age of behavioral assessment.

The effect of lead on IQ was estimated in all children using lifetime average, peak, concurrent, and average in infancy (6-24 months) blood lead levels. Lead effects on IQ for the subgroup of children whose peak lead concentration never exceeded 10 μg/dL also was estimated. Table 6-3.1 shows the covariate-adjusted changes in IQ for each 1 μg/dL increase in blood lead concentration for all children and children with peak blood lead concentrations below 10 μg/dL. In all cases, the effect estimates were larger in the subsample of children with peak blood lead concentrations below 10 μg/dL. For example, the overall estimate including all children indicated that an increase in the lifetime average blood lead concentration of 1 μg/dL was associated with a decrease of 0.46 points (95% CI: 0.15, 0.76) in IQ. In comparison, a 1 μg/dL increase in lifetime average lead concentration was associated with a decline of 1.37 points (95% CI: 0.17, 2.56) in children with peak blood lead concentrations below 10 μg/dL. In an accompanying editorial of the Canfield et al. (2003a) study, Rogan and Ware (2003) noted that the steepness in the concentration-response relationship below 10 μg/dL might have been influenced by 10 children with blood lead concentrations at or below 5 μg/dL and IQs above 115. However, they added that it was unlikely that the associations reported by Canfield et al. were solely due to these values. Regression diagnostics performed by Canfield et al. identified only one potential outlier (a child who had a low IQ and low lead concentration); however, this value was retained in all analyses as it did not pass the discordancy test.

In the Rochester study, the relationship between children's IQ score and their blood lead level was found to be nonlinear. A semiparametric analysis indicated a decline of IQ of

Table 6-3.1. Covariate-Adjusted Changes in IQ for Each 1 µg/dL Increase in Blood Lead Concentration[a]

Type of Blood Lead Measurement	n	At 3 Years of Age		At 5 Years of Age		Overall	
		β (95% CI)	p	β (95% CI)	p	B (95% CI)	p
All Children							
Lifetime average	172	−0.35 (−0.69, 0.00)	0.05	−0.57 (−0.93, −0.20)	0.003	−0.46 (−0.76, −0.15)	0.004
Peak	172	−0.19 (−0.39, 0.01)	0.06	−0.26 (−0.47, −0.05)	0.02	−0.23 (−0.40, −0.05)	0.01
Concurrent	171	−0.31 (−0.60, −0.01)	0.04	−0.61 (−0.99, −0.24)	<0.001	−0.46 (−0.74, −0.18)	0.002
Average in infancy (6-24 mo)	172	−0.32 (−0.71, 0.07)	0.10	−0.53 (−0.93, −0.13)	0.01	−0.43 (−0.77, −0.09)	0.02
Children with Peak Blood Lead Concentrations below 10 µg/dL[b]							
Lifetime average	101	−1.22 (−2.53, 0.09)	0.07	−1.52 (−2.94, −0.09)	0.04	−1.37 (−2.56, −0.17)	0.03
Peak	101	−1.36 (−2.46, −0.27)	0.002	−1.44 (−2.55, − 0.33)	0.01	−1.40 (−2.37, −0.44)	0.005
Concurrent	101	−1.36 (−2.37, −0.35)	0.009	−1.79 (−3.00, −0.60)	0.004	−1.58 (−2.50, −0.65)	0.001
Average in infancy (6-24 mo)	105	−0.58 (−1.75, 0.59)	0.32	−0.92 (−2.09, 0.25)	0.12	−0.75 (−1.78, 0.28)	0.15

[a] Estimates were adjusted for maternal IQ, race, level of education, use of tobacco during pregnancy, household income, HOME score, child's gender, birth weight, and iron status.

[b] A total of 71 children were found to have a peak blood lead concentration below 10 µg/dL at both ages; an additional 15 children had a peak concentration below 10 µg/dL at 3 years of age but at 5 years of age had a higher concentration or were not tested, and another 15 children had a peak concentration below 10 µg/dL at 5 years but were not tested at 3 years. The total number of children in the analysis of the average concentration in infancy is 105, because in 4 children the peak blood lead concentration occurred after the age of 24 months.

Source: Canfield et al. (2003a).

7.4 points for a lifetime average blood lead concentration of up to 10 µg/dL, while for levels between 10 to 30 µg/dL a more gradual decrease of approximately 2.5 points IQ was estimated. The authors concluded that the most important aspect of their findings was that effects below 10 µg/dL that have been observed in previous cross-sectional studies (e.g., Chiodo et al., 2004; Fulton et al., 1987; Lanphear et al., 2000; see Section 6.3.2.1.2) have been confirmed in this rigorous prospective longitudinal investigation.

Pooled-Analyses of Prospective Longitudinal Cohort Studies

Investigators have collectively analyzed the results of multiple independent studies using the methods of meta- and pooled data analyses. A powerful approach involves pooling the raw data from several high quality studies to examine concentration-response relationships in a large sample of children with diverse sociodemographic backgrounds and levels of exposure. The studies reviewed here are summarized in Annex Table AX6-3.2.

Lanphear et al. (2005) reported on a pooled analysis of seven prospective studies that were initiated prior to 1995. The analysis involved 1,333 children with complete data on confounding factors that were essential in the multivariable analyses. The participating sites included Boston, MA; Cincinnati, OH; Cleveland, OH; Rochester, NY; Mexico City; Port Pirie, Australia; and Kosovo, Yugoslavia. A prospective cohort study conducted in Sydney, Australia was not included because the authors were unable to contact the investigators (Cooney et al., 1989b, 1991). The sample size of 175 for children at age 7 years in the Sydney cohort and the wide confidence intervals of the effect estimates, as implied by the lack of significant associations, indicate that the nonavailability of this study is unlikely to influence the results of the pooled analysis by Lanphear et al.

The primary outcome measure was full scale IQ measured at school age (mean age at IQ testing was 6.9 years). All children were assessed with an age-appropriate version of the Wechsler scales. Four measures of lead exposure were examined: concurrent blood lead (blood lead level closest in time to the IQ test), maximum blood lead level (peak blood lead measured at any time prior to the IQ test), average lifetime blood lead (mean blood lead from 6 months to the concurrent blood lead test), and early childhood blood lead (defined as the mean blood lead from 6 to 24 months). A pooled analysis of the relationship between cord blood lead levels and IQ also was conducted in the subsample for which cord blood lead tests were available.

Multivariate regression models were developed adjusting for site as well as ten common covariates assessing factors likely to be confounders of the relationship between lead and cognitive development, including HOME scores, birth weight, maternal education and IQ, and prenatal substance abuse. A thorough statistical analytic strategy was employed to determine the linearity or nonlinearity of the relationship between blood lead levels and full-scale IQ. Regression diagnostics also were performed to ascertain whether lead coefficients were affected by collinearity or influential observations. The fit of all four measures of postnatal blood lead levels was compared using the magnitude of the model R^2. The blood lead measure with the largest R^2 (adjusted for the same covariates) was nominated a priori as the preferred blood lead index relating lead exposure to IQ in subsequent inspections of the relationships. Results were evaluated by applying a random-effects model (with sites random) rather than a fixed-effects model. The authors also examined the impact of any one site on the overall model by calculating the blood lead coefficient in seven identical models, each omitting one of the seven prospective cohort studies. Similar models were fitted for verbal and performance IQ as well.

The median lifetime average blood lead concentration was 12.4 µg/dL (5th-95th percentile 4.1-34.8) with about 18% of the children having peak blood lead levels below 10 µg/dL. The 5th to 95th percentile concurrent blood lead levels ranged from 0.8 to 4.7 µg/dL in the individual studies. The mean IQ of all children was 93.2 (SD 19.2) but this varied greatly between studies. All four measures of postnatal exposure were highly correlated. However, the concurrent blood lead level exhibited the strongest relationship with IQ, as assessed by R^2. Nevertheless, the results of the regression analyses for all blood lead measures were very similar. Multivariable analysis resulted in a six-term model including log of concurrent blood lead, study site, maternal IQ, HOME Inventory, birth weight, and maternal education. As illustrated in Figures 6-3.2 and 6-3.3, the shape of the log-linear model and the spline function indicated that the steepest declines in IQ were at blood lead concentrations below 10 µg/dL. The log-linear model estimated a decrement of 1.9 points (95% CI: 1.2, 2.6) in full scale IQ for a doubling of concurrent blood lead. Due to the log-linear relationship, the slope of the lead effect on IQ was greater in the lower ranges of exposure. The IQ point decrements associated with an increase in blood lead from <1 to 10 µg/dL compared to 10 to 20 µg/dL were 6.2 (95% CI: 3.8, 8.6) versus 1.9 (95% CI: 1.2, 2.6).

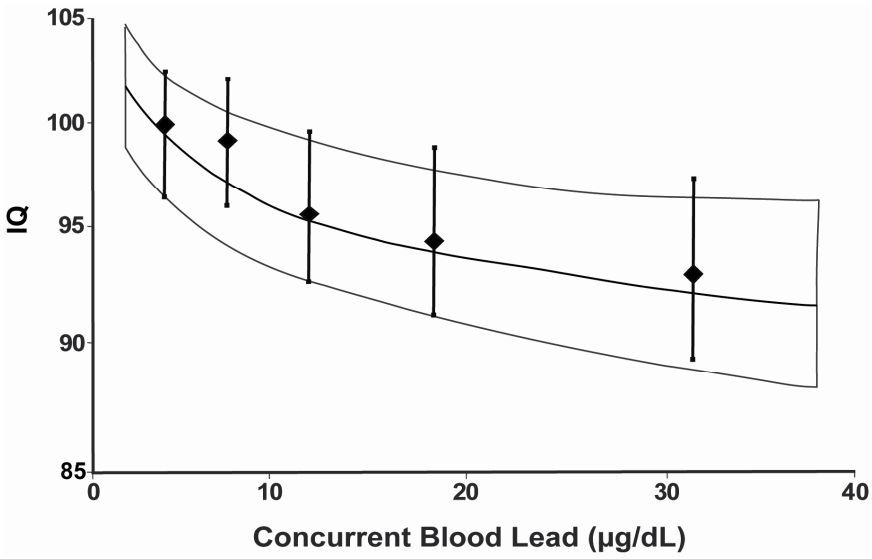

Figure 6-3.2. Log-linear model (95% CI shaded) for concurrent blood lead concentration adjusted for HOME score, maternal education, maternal IQ, and birth weight. The mean IQ (95% CI) for the intervals <5, 5-10, 10-15, 15-20, and >20 µg/dL are shown.

Source: Lanphear et al. (2005).

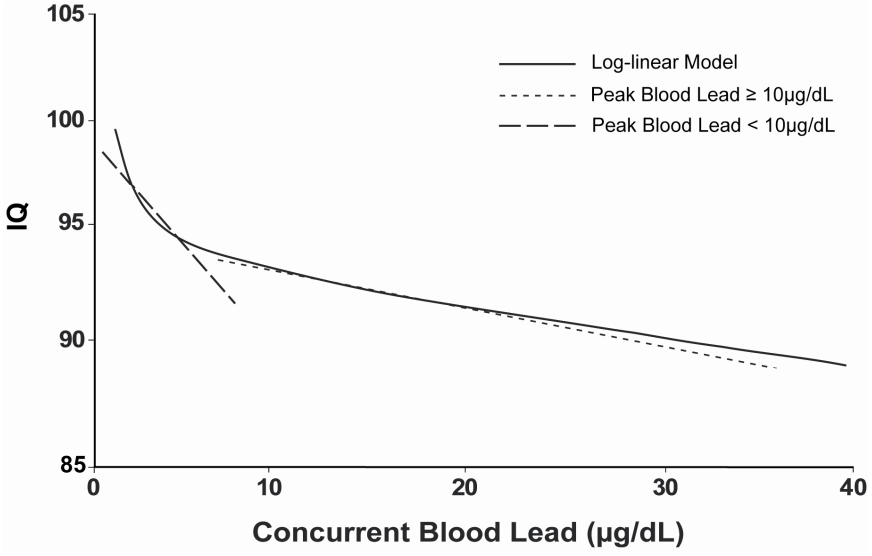

Figure 6-3.3. Log-linear model for concurrent blood lead concentration along with linear models for concurrent blood lead levels among children with peak blood lead levels above and below 10 µg/dL.

Source: Lanphear et al. (2005).

Rothenberg and Rothenberg (2005) reanalyzed the Lanphear et al. (2005) pooled study to examine the form of the concentration-response function for the lead exposure effect on child IQ. This further analysis also focused on concurrent blood lead levels. Rothenberg and Rothenberg reported that a log-linear relationship between blood lead and IQ was a significantly better fit within the ranges of the blood lead levels than was a linear-linear relationship ($p = 0.009$), with little evidence of residual confounding from included model variables. However, a segmented linear model also offers an appropriate alternative since limited data is available at lower levels.

The log-linear model in Lanphear et al. estimated a decline of 6.2 points in full scale IQ for an increase in concurrent blood lead levels from <1 to 10 μg/dL. This effect estimate was comparable to the 7.4 point decrement in IQ for an increase in lifetime mean blood lead levels up to 10 μg/dL observed in the Rochester study (Canfield et al., 2003a), as well as other studies reviewed above.

6.3.2.1.2 Cross-sectional Studies of Neurocognitive Ability

Among the cross-sectional studies reviewed in the 1986 Lead AQCD and the 1990 Supplement, the most thorough and methodologically rigorous were those of Needleman et al. (1979) and Fulton et al. (1987). Needleman et al. (1979) measured lead in the dentin of deciduous teeth in elementary school children from two Boston area communities. After statistical adjustment for a number of potential confounding factors, children in the higher tooth lead group performed significantly less well on full scale and verbal IQ. Differences in full scale IQ between the high and low tooth lead groups was on the order of 4.5 points.

The general population study by Fulton et al. (1987) studied 501 children aged 6-9 years in Edinburgh, Scotland who were at risk for lead exposure owing to a plumbosolvent water supply and a large number of houses with lead plumbing. Blood lead levels averaged 11.5 μg/dL (range 3-34). Following covariate adjustment, there were statistically significant relationships between concurrent blood lead levels and total scores on the British Ability Scale and the Quantitative and Reading subscales. Data showed a clear concentration-response relationship with no evidence of a threshold.

Recent cross-sectional studies of neurocognitive ability are summarized in Annex Table AX6-3.3. Key studies are further discussed in this section. Lanphear et al. (2000) examined the relationship between blood lead concentrations and cognitive deficits in a

nationally representative sample of 4,853 children aged 6 to 16 years children who participated in the third National Health and Nutrition Examination Survey (NHANES III). The purpose of the study was to examine the relationship between low blood lead concentrations (especially those below 10 µg/dL) and two subtests of the WISC-R, Block Design (a measure of visual-spatial skills) and Digit Span (a measure of short-term and working memory). Academic achievement tests also were administered but are discussed in a later section. A number of potential confounders were assessed and included in multivariable analyses including gender, racial/ethnic background, child's serum ferritin level, serum cotinine level, region of country, marital status and education level of primary caregiver, and a poverty index ratio (the ratio of total family income, as reported by the adult informant, to the federal poverty level for the year of the interview). Other potential confounders such as in utero and postnatal exposure to tobacco smoke, birth weight, and admission to the neonatal intensive care unit were only available for children between 6 and 11 years of age. Therefore, the authors conducted a secondary analysis of the data on these children to verify that inclusion of these potentially important variables did not alter the findings of the main analysis using the larger sample.

The geometric mean blood lead concentration for children in the study sample was 1.9 µg/dL (SE 0.1). Only 2.1% of the NHANES III sample in this analysis had blood lead concentrations greater or equal to 10 µg/dL. In multivariate analyses, a significant covariate-adjusted relationship was found between blood lead level and scores on both WISC-R subtest for all children as well as among those children with blood lead levels <10 µg/dL. Blood lead concentration also was significantly associated with Block Design when the multivariate analysis was restricted to children with blood lead levels <7.5 µg/dL. For a 1 µg/dL increase in blood lead level, Block Design scores declined by 0.10 points (SE 0.04) for all children, 0.13 points (SE 0.06) for children with blood lead levels <10 µg/dL, and 0.11 points (SE 0.06) for children with blood lead levels <7.5 µg/dL. The authors concluded that deficits in intellectual functioning were associated with blood lead levels <10 µg/dL. While a large number of potential confounding factors were controlled in these analyses, interpretation of results must be tempered by the fact that no data on maternal IQ or direct observations of caretaking quality in the home were available. Furthermore, it is not clear whether the cognitive deficits observed were due to lead exposure that occurred during early childhood or a function of concurrent exposure.

Chiodo et al. (2004) studied the relationship between blood lead concentrations and IQ, assessed using WISC-III, in a sample of 237 African-American inner-city children from Detroit, MI at 7.5 years of age. This cohort was derived from a larger study of the effects of prenatal alcohol exposure on child development. However, approximately 83% of children for whom blood lead levels were obtained had either low or no gestational exposure to alcohol. Blood lead levels were low with a mean of 5.4 µg/dL (SD 3.3, range 1-25). Following covariate adjustment, there was a statistically significant association between blood lead concentrations and full scale, verbal and performance IQ, with the strongest relationship observed for performance IQ. Significant effects of lead on full scale and performance IQ were still evident at blood lead concentrations below 7.5 µg/dL. Nonparametric smoothing analyses confirmed that these effects were linear in nature.

Walkowiak et al. (1998) conducted a cross-sectional study examining the relationships of low-level lead and mercury exposure, and various measures of neurocognitive and neuromotor functioning in 384 children aged 6 years in three German cities. Lead was measured in blood at the time of testing and mercury burden was estimated from urine samples. As their measure of IQ, they administered two subtests of the German WISC, Vocabulary and Block Design. These subtests were treated separately as well as a summed index, which served as a surrogate for full scale IQ. Blood lead concentrations were low (geometric mean 4.3 µg/dL [95th percentile 8.9]). Following covariate-adjustment, Vocabulary and the combined index, but not Block Design, exhibited negative associations with blood lead of statistical or borderline statistical significance; no associations were observed for mercury. The authors concluded that these findings roughly correspond with those of other studies that find effects of lead exposure on measures of intelligence at blood lead concentrations below 10 µg/dL. However, they also caution that some important covariates and potential confounding variables were not measured, including parental IQ and home environment (e.g., HOME score).

Rabinowitz et al. (1991) studied the relationship between lead measured in shed deciduous teeth (central incisors) and psychometric intelligence in 443 children in grades 1 to 3 in Taiwan. Two of the primary schools included in the study were in proximity to primary lead smelters. The Ravens Colored Progressive Matrices (RCPM), a test of nonverbal reasoning that is widely used in studies of non-western populations because of its more culturally neutral properties, was administered. Studies on a subsample of 60 children residing near the lead

smelters revealed mean blood lead level of 13.0 µg/dL (SD 4.4). Scores on the RCPM were negatively correlated with tooth lead concentrations. In multivariate analyses, parental education was a particularly important predictor of RCPM scores, but tooth lead concentrations still significantly predicted lower scores on the RCPM in families occupying the lowest social strata and among female subjects.

Kordas et al. (2004) examined the relationship between lead exposure and various indices of psychometric intelligence in a cohort of 602 first grade children attending public schools in Torreon, a highly industrialized city in northern Mexico. This study investigated whether lead-associated deficits in intellectual attainment might be explained by correlated nutritional factors such as iron status, anemia, and growth. The mean blood lead concentration was 11.5 µg/dL (SD 6.1). Approximately half of the children had blood lead concentrations below 10 µg/dL and only 20% of the subjects had blood lead levels in excess of 15 µg/dL. Subjects were administered Spanish or Mexican versions of the Peabody Picture Vocabulary Test-Revised (PPVT-R), the Cognitive Abilities Test (CAT), and subtests of the WISC-R (Coding, Digit Span, and Arithmetic subtests). Letter and Number Sequencing tests (adapted from the Trail Making Test, Trails A) also were administered. Following adjustment for sociodemographic variables, anemia, iron status, and growth, higher blood lead levels were significantly associated with poorer performance on the PPVT, WISC-R Coding, and Number and Letter Sequencing. The authors concluded that lead's association with iron deficiency anemia or growth retardation could not explain the relationship between lead and cognitive performance. The authors acknowledged that a major limitation of their study is the lack of earlier measures of lead exposure and nutritional status, and information on potentially confounding variables such as parental intelligence and quality of caretaking in the home.

Bellinger et al. (2005) reported on a study of the relationship between blood lead levels and IQ in 55 children aged 4 to 14 years in Chennai, India. This is the first published study that has investigated neurodevelopmental morbidities associated with undue lead exposure in Indian children. Children were recruited from a rural primary school on the outskirts of the city. The mean blood lead concentration was 11.1 µg/dL (SD 5.6, range 2.5-38.3). The Binet-Kamath Intelligence test along with other measures of neurobehavior were administered. The covariate-adjusted blood lead coefficient was negative but nonsignificant, perhaps due to the small sample size and highly variable performance of subjects with the lowest blood lead concentrations.

For example, the mean IQ of children in the highest blood lead quartile was 95.6 with a SD of 13.3 compared to 102.0 with a larger SD of 22.5 for children in the lowest blood lead quartile.

The cross-sectional studies examining the effect of lead on neurocognitive abilities varied widely in study location, population, age of testing, and outcomes measured. Collectively, they generally concluded that blood or tooth lead levels were significantly associated with declines in intelligence and other neurocognitive outcomes. In addition, these associations were consistently observed in studies with mean blood lead levels <10 μg/dL.

6.3.2.1.3 Meta-Analyses of Studies of Neurocognitive Abilities

The meta-analyses of studies investigating the association between lead and neurocognitive abilities included results from both prospective cohort studies and cross-sectional studies. The studies reviewed here are summarized in Annex Table AX6-3.2. Needleman and Gatsonis (1990) conducted a meta-analysis of 12 studies that used multiple regression techniques to assess the relationship between lead levels in tissues (blood or teeth) while adjusting for potentially confounding variables. Studies were weighted based on sample sizes, which ranged from 75 to 724 children. The authors divided studies into two groups according to the type of tissue analyzed for lead (blood or teeth). Joint p-values and average effect sizes as measured by partial correlation coefficients were calculated using two different methods by Fisher and by Mosteller and Bush (Rosenthal, 1984). The joint p-values for the blood lead studies were <0.0001 for both methods while joint p-values of <0.0006 and <0.004 were obtained for tooth lead studies. The partial correlations ranged from −0.27 to −0.0003. Sensitivity analyses revealed that no single study was responsible for the significance of the final findings. The authors concluded that the hypothesis that lead lowers children's IQ at relatively low dose was strongly supported by their quantitative analysis.

Another meta-analysis conducted by Schwartz (1994) took a different approach. Only studies relating blood lead to IQ were chosen for quantitative review since the concentration of lead in the bloodstream is the only index of exposure that has been used as the basis for public health policy. Three longitudinal and four cross-sectional studies relating blood lead to IQ were examined. Furthermore, while the work of Needleman and Gatsonis (1990) essentially involved combining partial correlations, the measure of effect used in the Schwartz analysis was the predicted change in full scale IQ as blood lead increased from 10 to 20 μg/dL. For the

prospective longitudinal studies, blood lead levels at 2 years of age or average blood lead levels up to 3 years of age were selected for the analysis. This approach by Schwartz may be related to the belief at the time of the analysis that blood lead levels during the first 3 years of life were the most critical in determining the severity of neurodevelopmental toxicity. The exclusion of blood lead levels from other time points may be of issue as it appears that later blood lead levels may be more predictive of mental deficits (Baghurst et al., 1992; Canfield et al., 2003a; Chen et al., 2005; Dietrich et al., 1993a; Factor-Litvak et al., 1993). Studies were weighted by the inverse of the variances using a random-effects modeling procedure. The estimated decrease in IQ for an increase in blood lead from 10 to 20 μg/dL was 2.6 points (95% CI: 1.8, 3.4). Sensitivity analyses indicated that the results were not determined by any individual study. Effect estimates were similar for longitudinal and cross-sectional studies. In another analysis, studies with mean blood lead concentrations below 15 μg/dL and above 15 μg/dL had estimated effect sizes of −3.23 points (95% CI: −5.70, −0.76) and −2.32 points (95% CI: −3.10, −1.54), respectively. When the study with the lowest mean blood lead level was examined in greater detail using nonparametric smoothing, no evidence of a threshold was observed down to a blood lead level of 1 μg/dL.

Pocock et al. (1994) conducted a review of the epidemiologic evidence for lead effects on IQ that included a meta-analysis. For the meta-analysis, the fixed-effect method described by Thompson and Pocock (1992) was used. Five prospective and 14 cross-sectional studies (with both tooth and blood lead measures) were included. For consistency, only blood lead levels at or around 2 years of age were considered for the prospective studies. Their overall conclusion was that a doubling of blood lead levels from 10 to 20 μg/dL, or tooth lead from 5 to 10 μg/g was associated with an average estimated deficit in IQ of around 1 to 2 points.

Other earlier meta-analyses of lead-IQ studies have been published but are not reviewed here, because later work greatly extended these efforts and included more studies, rendering these analyses outdated (Needleman and Bellinger, 1988; Schwartz, 1985; Thacker et al., 1992). The meta-analyses of studies investigating the effect of lead on neurocognitive ability consistently observed significant associations between blood or tooth lead levels and decrements in IQ. The analysis by Schwartz (1994) observed no evidence of a threshold at blood lead levels below 10 μg/dL.

6.3.2.2 Measures of Academic Achievement

There are relatively little data on the relationship between lead exposure and objective measures of academic achievement. A few earlier studies reported an inverse relationship between lead exposure and reading skills (Fergusson et al., 1988a; Fulton et al., 1987; Yule et al., 1981). Since the 1990 Supplement, more studies have focused on the practical consequences of childhood lead exposure by including measures of academic performance in their batteries. Studies reviewed in this section are summarized in Annex Table AX6-3.4.

Lanphear et al. (2000) examined the relationship between blood lead levels and a standardized measure of academic achievement in 4,853 children aged 6 to 16 years. The source of data for this study was the third National Health and Nutrition Examination Survey (NHANES III). This cohort was previously described in Section 6.3.2.1.2. Subjects were administered the Arithmetic and Reading subtests of the Wide Range Achievement Test-Revised (WRAT-R). The WRAT-R Arithmetic subtest includes oral and written problems ranging in level from simple addition to calculus, while the Reading subtest assesses letter recognition and word reading skills. The geometric mean blood lead concentration was 1.9 μg/dL. Only 2.1% of the subjects had blood lead levels equal to or greater than 10 μg/dL. Multiple linear regression revealed a 0.70 point (95% CI: 0.37, 1.03) decrement in arithmetic scores and a 0.99 point (95% CI: 0.62, 1.36) decrement in Reading scores for each 1 μg/dL increase in blood lead concentration ($p < 0.001$). In the next phase of the analysis, the adjusted relationship between performance on WRAT subtests and blood lead concentration for children with blood lead concentrations <10 μg/dL, <7.5 μg/dL, <5 μg/dL, or <2.5 μg/dL were carried out. Statistically significant inverse relationships between blood lead levels and performance for both Reading and Arithmetic subtests were found for children with blood lead concentrations below 5 μg/dL. Secondary analysis limited to younger children with data on all covariates did not alter findings from the main analysis. The authors concluded that results of these analyses suggest that deficits in academic skills are associated with blood lead concentrations lower than 5 μg/dL. However, although the relationship of blood lead concentration and achievement was adjusted for numerous potential confounders, the study lacked information on at least two covariates that have been shown to be important in other lead studies (HOME scores and parental IQ). Failure to adjust for these variables may have underestimated or overestimated the deficits in academic skills related to lead. Furthermore, as with all cross-sectional studies utilizing blood lead as the

index of dose, it is not clear whether the deficits in academic skills were due to lead exposure that occurred sometime during early childhood or due to concurrent exposure. Nevertheless, concurrent blood lead levels likely reflect both ongoing exposure and preexisting body burden.

Needleman et al. (1990) reexamined the Chelsea and Somerville, MA cohort of first and second graders recruited in the 1970s (Needleman et al., 1979). One hundred and thirty-two of the original 270 children were recalled. Neurobehavioral deficits in relationship to the concentration of lead in shed deciduous teeth had persisted into late adolescence. Subjects with dentin lead levels >20 ppm were at higher risk of dropping out of high school (adjusted odds ratio of 7.4, [95% CI: 1.4, 40.7]) and of having a reading disability (adjusted odds ratio of 5.8 [95% CI: 1.7, 19.7]). Higher dentin lead levels also were significantly associated with lower class standing, increased absenteeism, and lower vocabulary and grammatical reasoning scores on the Neurobehavioral Evaluation System (NES). The authors concluded that undue exposure to lead had enduring and important effects on objective parameters of success in real life.

Bellinger et al. (1992) administered a battery of neuropsychological tests to 148 children in the Boston Lead Study cohort at age 10 years. The authors administered the short-form of the Kaufman Test of Educational Achievement (KTEA) in addition to IQ studies. The KTEA assesses reading, math, and spelling skills. The primary outcome was the Battery Composite Score. As previously indicated, exposures in this cohort were low with a peak mean blood lead at 18 months of only 7.8 μg/dL (SD 5.7). The cohort had a high SES that consisted of white intact families with college-educated parents. Average KTEA scores in this cohort were approximately one standard deviation above the population mean. Nevertheless, postnatal blood lead levels measured at virtually all ages were significantly associated with lower KTEA Battery Composite Scores. However, after covariate-adjustment, including full scale IQ in the model, only blood lead levels at 24 months of age were significantly predictive of lower academic achievement. Over the range of approximately 0 to 25 μg/dL, Battery Composite scores declined by approximately 8.9 points (95% CI: 4.2, 13.6) for each 10 μg/dL increase in 24-month blood lead. The specific subscales of the KTEA that were most significantly associated with lead were Spelling and Math. Within the Math subscale, lead appeared to be more strongly associated with performance on the advanced quantitative Concepts/Applications items than on computation. The associations between these early measures of low level exposure to lead and achievement were significant even after adjustment for IQ, suggesting that

lead-sensitive neuropsychological processing and learning factors not reflected in indices of global intelligence may contribute to reduced performance on academic tasks.

Leviton et al. (1993) reported on the relationship between pre- and postnatal lead exposure and academic problems in approximately 2,000 children born in one Boston hospital between 1979 and 1980 using the Boston Teacher Questionnaire (BTQ). A teacher provided an assessment of each child's academic functioning when the child reached the age of 8 years. Mean umbilical cord blood lead was 6.8 μg/dL and mean tooth (dentin) lead concentration was 2.8 μg/g. There was limited information on covariate factors. However, following adjustment for potential confounding variables, elevated dentin lead concentrations were associated with statistically significant reading and spelling difficulties as assessed by the BTQ among girls. The authors concluded that their findings supported the case for lead-associated learning problems at levels prevalent in the general population. However, they added that the inability to assess child-rearing quality in this questionnaire study conducted by mail limits the inferences that came be drawn from the findings.

Fergusson et al. (1993) examined the relationship between dentin lead levels in shed deciduous teeth at 6-8 years and measures of academic attainment and classroom performance in a birth cohort of over 1,200 New Zealand children enrolled in the Christchurch Health and Development Study when they reached 12-13 years of age. This study was an extension of earlier work in these children indicating a relationship between low lead levels and deficits in academic skills around the age of 8 years (Fergusson et al., 1988a). Average dentine lead levels in the cohort were 6.2 μg/g (SD 6.2). Measures of academic performance included word recognition from the Burt Reading Test, reading comprehension from the Progressive Achievement Test, a general measure of scholastic skills based on children's scores on the Test of Scholastic Abilities, and teacher ratings of classroom performance in the areas of reading, written expression, and mathematics. Following adjustment for a wide range of covariates (including residence in potentially lead-hazardous housing), dentin lead levels were significantly associated with virtually every formal index of academic skills and teacher ratings of classroom performance. Statistical treatment of the data included a multivariate analysis of all 12 regression equations simultaneously using LISREL modeling methods. This conservative analysis clearly showed that the probability of observing these results under the null hypotheses that lead was unrelated to all covariate-adjusted test outcomes was extremely small. In an

adjunct analysis, Fergusson and Horwood (1993) examined the effects of low-level lead exposure on the growth of word recognition in this cohort from 8 to 12 years of age. The New Zealand data were analyzed using growth curve modeling methods. After adjustment for potential confounding variables, children with dentin lead levels equal to or greater than 8 μg/g displayed significantly slower growth in word recognition abilities with no evidence of catch up. The authors concluded that these results were consistent with their earlier analyses and suggest that early exposure to very low levels of lead result in small but detectable and enduring deficits in children's cognitive abilities.

Academic achievement in relationship to lead was reexamined in the New Zealand cohort when subjects reached 18 years of age (Fergusson et al., 1997). The sample at 18 years consisted of 881 subjects, or approximately 70% of the original cohort. Measures of educational achievement included the Burt Reading Test, number of years of secondary education, mean number of School Certificate passes (based on results of national examinations), and leaving school without formal qualifications (analogous to failure to graduate from high school in the U.S.). As in previous analyses, a wide range of potentially confounding socioherediatary factors were measured and controlled for in multivariable analyses, which included both linear and logistic regressions. Prior to and following covariate adjustment there were statistically significant concentration-response relationships between dentin lead concentrations and lower reading test scores, having a reading level of less than 12 years, failing to complete 3 years of high school, leaving school without qualifications, and mean number of School Certificates subjects passed. The authors conclude that their results are consistent with the view that there is a relationship between early exposure to low levels of lead and later educational outcomes. The late results of the New Zealand studies confirm the findings of Needleman et al. (1990) in a cohort with lower levels of exposure to environmental lead.

Rabinowitz et al. (1992) examined the relationship between tooth lead concentrations and scores on BTQ clusters in 493 Taiwanese children in grades one through three. Mean lead levels in incisors were 4.6 μg/g (SD 3.5). Factors associated with lead and the BTQ included 13 variables measuring perinatal, familial, and economic parameters. Prior to adjustment for covariates, girls in this sample with higher exposures to lead evinced a borderline significant trend for reading difficulties while boys displayed significantly increased difficulties with respect to activity levels and task attentiveness. In multiple logistic regression models, the tooth lead

terms failed to achieve statistical significance. The authors concluded that lead levels found in the teeth of children in their Taiwanese sample were not associated with learning problems or syndromes as assessed by the BTQ.

Wang et al. (2002) examined the relationship between blood lead levels and class ranking in 934 third graders living in an urban industrial area of Taiwan. The outcome variables were grades for Chinese (reading and writing), Mathematics, History and Society, and Natural Science. To avoid the impact of teacher's bias in grading criteria, the authors converted the children's grades into class rankings. A limited number of potentially confounding factors were measured, including maternal education and father's SES. Mean blood lead level was 5.5 µg/dL (SD 1.89). In multiple regression analyses adjusting for gender, maternal education, and father's SES, blood lead was significantly associated with lower class ranking in all academic subjects. The major shortcoming of this cross-sectional study is the lack of control for potentially important confounding factors such as parental intelligence. However, the strength and consistency of the reported relationships suggest that relatively low levels of lead may play a role in lowering academic performance.

The results of these studies strongly suggest that lead exposure plays a role in the academic performance of children. The effects of lead on academic achievement appear to include children with blood lead levels that do not exceed 10 µg/dL.

6.3.2.3 Measures of Specific Cognitive Abilities

Outcomes of specific cognitive abilities, in particular, the domains of Attention and Executive Functions, Language, Memory and Learning, and Visuospatial Processing have been examined in some detail in recent studies. These studies are summarized in Annex Table AX6-3.5.

In the aggregate, studies suggest that lead exposure impairs a child's ability to regulate attention and engage several related higher order cognitive processes that have come to be termed "executive functions." Executive functions refer to strategic planning, control of impulses, organized search, flexibility of thought and action, and self-monitoring of one's own behavior—activities that help the subject maintain an appropriate mental set in order to achieve an immediate or future goal (Spreen et al., 1995). In some earlier studies, increased lead exposure was found to be associated with a higher frequency of negative ratings by teachers

and/or parents on behaviors such as inattentiveness, impulsivity, distractibility, and impersistence in assigned tasks, as well as slow psychomotor responses and more errors on simple, serial, and choice reaction time tasks (e.g., Hatzakis et al., 1989; Hunter et al., 1985; Needleman et al., 1979; Raab et al., 1990; Winneke et al., 1990). The concept that lead may impact executive functions in particular is biologically plausible. The prefrontal cortex is highly innervated by projections of neurons from the midbrain and has the highest concentration of dopamine of all cortical areas. Dopamine plays a key role in cognitive abilities mediated by the prefrontal cortex. It has been known for some time that the dopamine system is particularly sensitive to lead based upon data from studies of rodents and nonhuman primates (Cory-Slechta, 1995).

Bellinger et al. (1994) examined a portion of the original Chelsea and Somerville cohorts at 19-20 years of age. The principal neurobehavioral outcomes in the investigation were scores on a battery of attentional measures assembled by Mirsky (1987). Higher tooth lead concentrations were significantly associated with poorer scores on the Focus-Execute and Shift factors of the battery leading the authors to conclude that early lead exposure may be associated with poorer performance on executive/regulatory functions, which are thought to depend on the frontal or prefrontal regions of the brain.

Stiles and Bellinger (1993) administered a neuropsychological battery of tests to 10-year-olds in the Boston Lead Study cohort. A large number of assessments were made and, as the authors acknowledge, the number of significant associations was about equal to those that would be expected by chance. However, as in previous studies, tasks that assess attentional behaviors and executive functions tended to be among those for which lead was a significant predictor of performance. For example, higher blood lead concentrations at 2 years were significantly associated with lower scores on the Freedom from Distractibility factor of the Wechsler scales and an increase in the percentage of preservative errors on the Wisconsin Card Sorting Test and the California Verbal Learning Test.

Canfield et al. (2003b) conducted a comprehensive examination of the relationship between low-level lead exposure, executive functioning, and learning in children from the Rochester Lead Study cohort at 48 and 54 months of age. The authors used the Shape School Task (Espy, 1997), which requires only knowing simple shape and primary color names. However, embedded in the tasks are protocols requiring inhibition, attention switching, and a

combination of inhibition and switching mental sets. Following covariate-adjustment, blood lead level at 48 months was negatively associated with children's focused attention while performing the tasks, efficiency at naming colors, and inhibition of automatic responding. Children with higher blood lead concentrations also completed fewer phases of the task and knew fewer color and shape names.

Canfield et al. (2004) also administered portions of the Cambridge Neuropsychological Testing Automated Battery (CANTAB) to 174 children in the Rochester cohort at approximately 66 months. Children were tested with the Working Memory and Planning CANTAB assessment protocols to assess mnemonic and executive functions. Blood lead levels ranged from 0-20 μg/dL in this cohort. Following covariate adjustment, children with higher blood lead levels showed impaired performance on tests of spatial working memory, spatial memory span, cognitive and cognitive flexibility, and planning as indexed by tests of intradimensional and extradimensional shifts and an analog of the Tower of London task.

Ris et al. (2004) administered an extensive neuropsychological battery to 16-17 year old subjects from the Cincinnati Lead Study cohort. In addition to executive functions as assessed by the Wisconsin Card Sorting Test and the Rey-Osterrieth Complex Figure, other domains examined included attention, memory, achievement, verbal abilities, visuoconstructional skills, and fine-motor coordination. A factor analysis of scores selected a priori revealed five factors that included Attention. A strong "executive functions" factor did not emerge. Following covariate-adjustment, the strongest associations between lead exposure and performance were observed for factor scores derived from the Attention component, which included high loadings on variables from the Conners Continuous Performance Test. However, this relationship was restricted to males as indicated by a strong lead by gender interaction. This obtained gender interaction suggests that neuromechanisms sub-serving attention were affected by lead in this cohort for boys but not girls. This is not surprising given the heightened vulnerability of males for a wide range of developmental perturbations. A substantial gender difference in the incidence of Attention Deficit/Hyperactivity Disorder (ADHD) is well established, and one could speculate that early exposure to lead exacerbates a latent potential for such problems.

Visual-spatial skills have also been also been explored in some depth by a few studies. When investigations of lead-exposed children have used global IQ measures and conducted subscale analyses, it has been observed that Performance IQ or subtests contributing to the

performance IQ (i.e., Block Design) are frequently among the most strongly associated with biological indices of exposure (Baghurst et al., 1992; Chiodo et al., 2004; Dietrich et al., 1993a; McMichael et al., 1988; Wasserman et al., 1994). Dietrich et al. (1991, 1992) have also observed that integrated measures of lead exposure over a child's lifetime are most consistently associated with simultaneous processing abilities, cognitive functions closely associated with visual-spatial integration skills and right cerebral functioning (Kaufman and Kaufman, 1983). In addition, studies employing specific measures of visual-motor integration skills such as the Developmental Test of Visual Motor Integration (VMI), the Bender Visual-Motor Gestalt Test and other have found them to be among the most consistently associated with early exposure to lead (Baghurst et al., 1995; Dietrich et al., 1993b; Wasserman et al., 2000a; Winneke et al., 1990). In a follow-up of subjects in the Cincinnati Lead Study cohort at 16 years, Ris et al. (2004) observed a significant association between prenatal maternal blood lead levels and deficits in visual-spatial and constructional skills as indexed by Visual-Constructional factor scores. Variables with high loadings on this factor included scores on the WISC-III Block Design subtests and selected variables from the Rey Osterrieth Complex Figure.

However, it is still unclear whether the domains of attention/executive functions or visual-motor integration per se are specifically sensitive to lead. This is because there is rarely a one-to-one correspondence between performance on a focused neuropsychological test and an underlying neuropsychological process. Thus, for example, a low score on the Berry VMI may reflect singular or multiple neurobehavioral deficits, including difficulties with graphomotor control, visual perception, behavioral monitoring (impulsivity), or planning (executive functions).

6.3.2.4 Disturbances in Behavior, Mood, and Social Conduct

The effects of lead on behavior and mood of children has been an area of recent research. Studies conducted prior to 1990 clearly pointed to behavioral problems as potential sequelae of lower level lead toxicity in children. Several early case control studies linked lead to hyperactivity (David et al., 1972, 1976, 1979). Low levels of lead in blood and/or teeth have been associated with teacher ratings of hyperactive behavior, aggression, and attention problems (e.g., Fergusson et al., 1988b; Hatzakis et al., 1985; Silva et al., 1988; Thomson et al., 1989; Yule et al., 1984). In the seminal study by Needleman et al. (1979), children with higher

concentrations of lead in dentin were more likely to be rated unfavorably by teachers on the dimensions of hyperactivity, impulsivity, and frustration tolerance. New studies reviewed in this section are summarized in Annex Table AX6-3.6.

While there is no compelling evidence that lead is directly related to ADHD, elevated blood or tooth lead levels have been linked to behavioral features of ADHD, including distractibility, poor organization, lacking persistence in completing tasks, and daydreaming (Bellinger and Rappaport, 2002). Bellinger et al. (1994) studied the relationship between early exposure to lead and problem behaviors in the classroom in a cohort of 1,782 children born at one hospital in Boston. Lead levels in umbilical cord blood were low (mean 6.8 µg/dL [SD 3.1]) as were tooth lead levels (mean 3.4 µg/g [SD 2.4]). Teachers filled out the Achenbach Child Behavior Profile (ACBP) which yields both broad and narrow band scales indexing externalizing and internalizing problems. Cord blood lead levels were not associated with the prevalence or nature of behavioral problems reported by teachers. However, tooth lead level was significantly associated with ACBP Total Problem Behavior Scores (TPBS). TPBS scores increased by approximately 2 points for each log unit increase in tooth lead. Statistically significant tooth lead-associated increases in both externalizing and internalizing scores also were noted. Each log unit increase in tooth lead was associated with a 1.5 point increase in scores for these broadband scales assessing under- and overcontrol of behavior. Only weak associations were noted between tooth lead concentrations and the tendency to score in the clinically significant range on these scales. As the authors noted, it was somewhat surprising that lead exposure was not more strongly related to externalizing behavior problems than with internalizing behavior problems. This contradicted several earlier investigations, including one by Sciarillo et al. (1992) described below. It may be that more attention has been accorded under controlled behaviors, because they are more readily visible and disruptive in settings such as the classroom. Therefore, internalizing problems may be part of the full spectrum of behaviors in which lead's developmental neurotoxicity is expressed in children. The authors also cautioned that residual confounding could not be ruled out, because of the lack of covariate information on parental psychopathology or direct observations of the family environment—a problem not unique to this particular study. Nevertheless, these data are in accord with other studies that social and emotional dysfunction may be another expression of increased lead exposure during the early postnatal period.

Sciarillo et al. (1992) examined the relationship between early exposure to lead and child behavior in a cohort of 150 subjects in Baltimore, MD. Children were separated into high exposure (two consecutive blood lead concentrations greater than or equal to 15 µg/dL) and low exposure groups. Blood lead also was treated as a continuous variable in regression analyses. Mothers of 2-5 year old children were administered the Achenbach Child Behavior Checklist (CBCL) and given the Center for Epidemiologic Studies Depression scale (CESD) as a control measure. Mean blood lead concentrations were 28.6 µg/dL (SD 9.3) and 11.3 µg/dL (SD 4.3) in the high and low exposure groups, respectively. When compared to the lower exposed group, children with higher blood lead levels had a significantly higher mean TBPS, and internalizing and externalizing scores. Using regression procedures to control for maternal symptoms on the CESD, blood lead concentrations were still significantly associated with an increase in the TBPS. Children in the high exposure group were also nearly 3 times more likely to have a TBPS in the CBCL's clinical range. A significantly higher percentage of these children scored in the clinical range for CBCL subscales measuring aggressive and destructive behavioral tendencies.

Fergusson et al. (1993) examined the relationship between tooth lead levels and inattention/restlessness in the large national New Zealand study of over 1,000 children at 12 and 13 years of age. Mothers and teachers were asked to respond to a series of items derived from the Rutter and Conners parental and teacher questionnaires. The selected items related to the degree to which the child was restless, inattentive, easily distracted, and lacking in concentration. At each age, an index of the subject's propensity to inattentive and restless behavior was obtained by summing the total reports of attention deficit behaviors made by both teacher and parent respondents. Following adjustment for a wide range of sociodemographic and other covariate factors, a statistically significant, concentration-response relationship was observed between tooth lead concentrations (range 1-12+ µg/g) and the inattention/restlessness variable. The authors concluded that their results were consistent with the view that early mildly elevated lead levels were associated with small but long term deficits in attentional behaviors.

Two prospective studies have also examined measures of early exposure to lead and behavioral problems as assessed by the Achenbach system. Wasserman et al. (1998) studied the relationship between lead exposure and behavior in the Yugoslavian prospective study. The study survey 379 children at 3 years of age with the parent report form of the Achenbach CBCL. Following covariate adjustment, concurrent blood lead levels were significantly associated with

scores on the Destructive Behaviors CBCL subscale, although the variance accounted for by lead was small compared to sociodemographic factors. As blood lead increased from 10 to 20 µg/dL, CBCL subscale scores increased by approximately 0.5 points. The authors concluded that while statistically significant, the contribution of lead to social behavioral problems in this cohort was small compared to the effects of correlated social factors. Burns et al. (1999) examined the relationship between lead exposure and children's emotional and behavioral problems at ages 11-13 years in the Port Pirie, Australia cohort study. After adjusting for a number of confounding variables, including HOME scores, maternal psychopathology and the child's IQ, regression models showed that for an increase in average lifetime blood lead concentrations from 10 to 30 µg/dL, the externalizing behavior problem score increased by 3.5 points (95% CI: 1.6, 5.4) in boys, but only by 1.8 points (95% CI: –0.1, 11.1) in girls. In contrast, internalizing behavior problems were predicted to increase by 2.1 points (95% CI: 0.0, 4.2) in girls, but by only 0.8 points (95% CI: –0.9, 2.4) in boys.

Recently, the question of lead's role in delinquent and criminal behavior has been addressed in several investigations. Previous studies linking attention deficits, aggressive and disruptive behaviors, and poor self-regulation with lead have raised the prospect that early exposure may result in an increased likelihood of engaging in antisocial behaviors in later life.

Denno (1990) surveyed 987 Philadelphia African American youths enrolled in the Collaborative Perinatal Project. Data were available from birth through 22 years of age. The analysis initially considered over 100 predictors of violent and chronic delinquent behavior. Repeat offenders presented consistent features such as low maternal education, prolonged male-provider unemployment, frequent moves, and higher lead intoxication (although Denno does not indicate the level of lead intoxication in her report). In male subjects, a history of lead poisoning was among the most significant predictors of delinquency and adult criminality.

Needleman et al. (1996) examined the relationship between lead exposure and several measures of behavioral disturbance and delinquent behavior in subjects from the Pittsburgh Youth Study. The Pittsburgh Youth Study is a prospective study of the developmental course of delinquency (Loeber et al., 1991). The population consisted of 850 boys who were prescreened with an instrument that measured serious and potentially indictable behaviors extracted from the teachers' and parents' CBCL. Subjects who scored above the 30th percentile on the risk score and an approximately equal number of subjects randomly selected from the remainder of the

distribution formed the sample (n = 503). Body burden of lead was measured in the tibia by K-shell XRF. Measures of antisocial behavior were administered at 7 and 11 years of age and included the Self Reported Antisocial Behavior scale (SRA), the Self Report of Delinquent Behavior (SRD), and the parents' and teachers' versions of the CBCL. Outcome data were adjusted for a number of covariates including mother's IQ, SES, childhood medical problems, and quality of child rearing. Parents of subjects with higher lead levels in bone reported significantly more somatic complaints, more delinquent and aggressive behavior, and higher internalizing and externalizing scores. Teachers reported significant increases in scores on somatic complaints, anxious/depressed, social problems, attention problems, delinquent behavior, aggressive behavior, and internalizing and externalizing problems in the higher lead subjects. At 11 years, subjects SRD scores also were significantly related to bone lead levels. More of the high lead subjects had CBCL scores in the clinical range for the CBCL subscales assessing attention problems, aggression, and delinquency. Odds ratios for these outcomes ranged from 1.5 (95% CI: 0.45, 4.9) for parental reports of aggression to 19.5 (95% CI: 8.9, 41.6) for attention problems. The authors concluded that lead exposure was associated with an increased risk for antisocial and delinquent behavior.

Dietrich et al. (2001) reported on the relationship between early exposure to lead and juvenile delinquency in 195 subjects from the Cincinnati Lead Study. Subjects were between 16 and 17 years of age when examined. As previously described, this is an inner-city cohort of urban children exposed to relatively high levels of lead by virtue of their residence in older, deteriorated housing units. Relationships between prenatal (maternal) and postnatal exposure to lead (through serial blood lead determinations), and antisocial and delinquent behaviors (self- and parental reports) were examined. Parents were administered a questionnaire developed specifically for the study while CLS subjects were given the SRD. A wide range of candidate covariates and confounders were examined, but the only ones predicting antisocial or delinquent behavior were birth weight, HOME scores, SES, and parental IQ. In multiple linear regression analyses, prenatal exposure was significantly associated with a covariate-adjusted increase in the frequency of parent-reported delinquent and antisocial acts, while prenatal and postnatal exposure to lead was significantly associated with a covariate-adjusted increase in frequency of self-reported delinquent and antisocial behaviors, including marijuana use. To clarify the concentration-response relationships, blood lead indices were transformed to categorical

variables and least-square means were calculated from an analysis of covariance procedure. Subjects in the highest prenatal blood lead category (>10 µg/dL) engaged in 2.3 more delinquent acts over the preceding 12 months than subjects in the lowest category (≤5 µg/dL). Using average childhood blood lead levels, subjects in the medium (16-20 µg/dL) and highest (>20 µg/dL) category engaged in approximately 1.5 more delinquent acts compared to the lowest category (≤10 µg/dL). Subjects in the highest 78-month blood lead category (>15 µg/dL) engaged in 4.5 more delinquent acts than subjects in the lowest category (≤5 µg/dL). The authors concluded that lead might play a measurable role in the epigenesis of behavioral problems in inner-city children independent of other social and biomedical cofactors assessed in the study.

Needleman et al. (2002) conducted a case-control study where they examined the levels of lead in bone of 194 adjudicated delinquents and 146 non-delinquent community controls. Subjects were recruited from high schools in the city of Pittsburgh and environs of Allegheny County, PA. Since many delinquents are not arrested or adjudicated, care was taken to ensure that unidentified delinquents did not populate the control group. Potential control subjects were excluded from the analyses if they were found to have a Juvenile Court record or an SRD score above the 90th percentile. Tibial bone lead was measured by K-shell XRF. Covariates included race, parental education and occupation, presence of two parental figures in the home, number of children in the home, and neighborhood crime rate. Logistic regression analyses were undertaken to model the association between delinquent status and bone lead concentration. Cases had significantly higher average concentrations of lead in tibia than controls (11.0 µg/g [SD 32.7] versus 1.5 µg/g [SD 32.1]). Stratified analyses revealed this was true for both white and African American subjects. Following adjustment for covariates, adjudicated delinquents were four times more likely to have bone lead concentration greater than 25 µg/g than controls (odds ratio of 4.0 [95% CI: 1.4, 11.1]). The effect of lead on delinquency was found to be substantial in this study. Bone lead level was the second strongest factor in the logistic regression models, exceeded only by race. In models stratified by race, bone lead was exceeded as a risk factor only by single parent status. The authors concluded that elevated body lead burdens were associated with elevated risk for adjudicated delinquency.

The extension of lead effects into delinquent and criminal behavior is significant for both the individual and society as a whole. The particular biological mechanisms that may underlie

lead's effects on aggression, impulsivity, and poor self-regulation are not clearly understood. However, lead impacts a large number of sites and processes in the brain that are involved in impulse control (Lidsky and Schneider, 2003). Needleman et al. (2002) proposed another pathway. In addition to lead's direct impact on brain development and neuronal function, lead exposure may increase risk of delinquency through a separate, indirect route: impaired cognitive abilities and academic performance. In other words, students who have difficulties in school and fail to achieve academic goals are more likely to become lawbreakers.

6.3.2.5 Sensory Acuities

In comparison to cognitive outcomes, there has been relatively less interest in the effects of lead on sensory functions. However, there are clear indications that lead exposure during the developmental period has an impact on complex aspects of visual and auditory acuities. Much of this work has been carried out in animal models (Otto and Fox, 1993). Epidemiologic studies have typically assessed hearing thresholds and features of auditory processing in lead-exposed children. Studies reviewed in this section are summarized in Annex Table AX6-3.7.

Schwartz and Otto (1987) observed significant lead-associated elevations in pure-tone hearing thresholds at various frequencies within the range of human speech among over 4,500 4-19 year old subjects in NHANES II. In a later study, this finding was replicated in a sample of over 3,000 6-19 year old subjects in the Hispanic Health and Nutrition Examination Survey (HHANES) (Schwartz and Otto, 1991). An increase in blood lead from 6 to 18 µg/dL was associated with a 2 db loss in hearing at all frequencies, and an additional 15% of children had hearing thresholds that were below the standard at 2,000 Hz. These relationships continued at blood lead levels less than 10 µg/dL.

Dietrich et al. (1992) assessed the relationship between scores on a test of central auditory processing (SCAN) and prenatal/postnatal blood lead concentrations in 215 children 5 years of age drawn from the Cincinnati Lead Study. Higher prenatal, neonatal, and postnatal (up to concurrent) blood lead concentrations were associated with more incorrect identification of common monosyllabic words presented under conditions of filtering (muffling). Other variables associated with impaired central auditory processing included the results of pure-tone audiometry testing, social class, HOME scores, birth weight, gestational age, a measure of obstetrical complications, and consumption of alcohol during pregnancy. Following adjustment

for these covariates, neonatal and postnatal blood lead levels remained significantly associated with impaired performance on the Filtered Word subtest, more prominently in the right ear. In the right ear, the Filtered Word subtest score decreased by 0.7 points ($p < 0.05$; 95% CI not presented) for a 10 µg/dL increase in lifetime average blood lead levels.

Osman et al. (1999) examined the relationship between concurrent blood lead levels and hearing loss in 155 children 4-14 years of age living in an industrial region of Poland. Blood lead levels ranged from 1.9 to 28 µg/dL (median 7.2 µg/dL). Hearing thresholds increased significantly with higher blood lead levels at all frequencies (500-8,000 Hz). This relationship remained statistically significant when restricted to children with blood lead levels below 10 µg/dL.

A limited number of epidemiologic studies provide supportive evidence of a relationship between lead exposure and auditory processing. Lead-related deficits in hearing and auditory processing may be one plausible mechanism by which an increased lead burden might impede a child's learning (Bellinger, 1995).

6.3.2.6 Neuromotor Function

Relatively few studies have focused on neuromotor deficits as an outcome of early lead exposure. However, those that have examined motor functions in lead-exposed children often report positive findings. Studies reviewed in this section are summarized in Annex Table AX6-3.8.

In an early study, unsteadiness, clumsiness, and fine-motor dysfunctions were noted in a group of mildly symptomatic lead-poisoned children in Boston, with such effects persisting long after medical treatment (Pueschel et al., 1972). A study of moderately exposed children living in the vicinity of a longstanding lead smelter in Greece found that children with blood lead levels of 35-60 µg/dL had significantly lower scores on both the Gross and Fine Motor Composite scores from the Oseretsky scales when compared to controls (Benetou-Marantidou et al., 1988).

Only two modern prospective studies of lead have assessed motor development in a comprehensive manner. Dietrich et al. (1993b) investigated the association between lead exposure and motor developmental status in 245 children 6 years of age in the Cincinnati Lead Study cohort. Following covariate adjustment, they found that postnatal lead exposure was significantly associated with poorer scores on measures of bilateral coordination, visual-motor

control, upper-limb speed and dexterity, and the fine motor composite from the Bruininks-Oseretsky scales. Neonatal, but not prenatal, blood lead concentrations also were significantly associated with poorer scores on upper-limb speed and dexterity and the fine motor composite. The strongest and most consistent relationships were observed with concurrent blood lead levels (mean 10.1 µg/dL [SD 5.6]). A 10 µg/dL increase in concurrent blood lead levels was associated with a 4.6 point (95% CI: 2.1, 7.1) decline in the fine motor composite score. In the same Cincinnati cohort, postnatal lead exposure was associated with greater postural instability as assessed by a microprocessor-based strain gauge platform system (Bhattacharya et al., 1995). When assessed at 16 years of age, 78-month postnatal blood lead levels were significantly associated with poorer fine-motor skills as indexed by covariate-adjusted factor scores derived from a factor analysis of a comprehensive neuropsychological battery (Ris et al., 2004). The variables loading highly on the fine-motor component came from the grooved pegboard and finger tapping tasks.

Some results of the Cincinnati Lead Study were replicated by Wasserman et al. (2000a) in the Yugoslavian Prospective Study. The authors adapted the Bruininks-Oseretsky Test of Motor Proficiency for use in their population residing in two towns in the province of Kosovo. The measure of exposure was the log of the lifetime average blood lead concentration through 54 months of age. Following covariate-adjustment, average childhood blood lead concentrations were associated with poorer fine motor and visual motor function, but were found to be unrelated to gross motor function.

A recent study by Despres et al. (2005) of multiple exposures including lead, mercury, and polychlorinated biphenyls found that only blood lead concentrations measured at the time of assessment were associated with neuromotor functions in 110 preschool Inuit children residing in Canada. The mean blood lead level was 5.0 µg/dL (range 0.8-27.1). Blood lead levels were significantly associated with increased reaction time, sway oscillations, alternating arm movements, and action tremor. Ten percent of the children had blood lead levels greater than 101 µg/dL. After eliminating these children from the analyses, results remained significant for reaction time, sway oscillations, and alternating arm movements. These findings indicated that neuromotor effects of lead occurred at blood lead concentrations below 10 µg/dL.

6.3.2.7 Brain Anatomical Development and Activity

Electrophysiological evaluations have been conducted on lead-exposed children in attempts to obtain a more direct measure of the toxicant's impact on the nervous system. Much of this work was conducted by Otto and colleagues during the 1980s (e.g., Otto et al., 1985). Studies reviewed in this section are summarized in Annex Table AX6-3.9. These studies have demonstrated effects of lead on neurosensory functioning (auditory and visual evoked potentials) within a broad range of exposures (Otto and Fox, 1993).

Rothenberg et al. (1994) reported that higher maternal blood lead levels at 20 weeks of pregnancy were associated with increased I-V and III-V interpeak intervals in the brainstem auditory evoked response recorded in 1-month-old infants. Mean maternal blood lead level at 20 weeks in this subsample from the Mexico City Prospective Study was only 7.7 μg/dL with a range of 1-30.5 μg/dL. Rothenberg et al. (2000) repeated these measurements with a larger group of 5-7 year old children (n = 133). In contrast to their previous findings, prenatal blood lead levels at 20 weeks were associated with decreased interpeak intervals. However, after fitting a nonlinear model to their data, they observed that I-V and III-V interpeak intervals decreased as blood lead rose from 1 to 8 μg/dL and increased as blood lead rose from 8 to 30 μg/dL. The biphasic effect was only observed with maternal blood leads at 20 weeks of pregnancy. Increasing postnatal blood lead at 12 and 48 months was related to decreased conduction intervals for I-V and III-V interpeak intervals across the entire blood lead range.

The methods of Magnetic Resonance Imaging (MRI) and Magnetic Resonance Spectroscopy (MRS) have recently been applied in studies of lead-exposed children. Trope et al. (1998) were the first to apply MRI and MRS in an evaluation of a lead-exposed subject. The subject was a 10 year old boy who had a history of elevated blood lead levels as a toddler (e.g., 51 μg/dL at 38 months). The subject was compared to his 9-year-old unexposed cousin. The investigation was particularly focused on *N*-acetylaspartate, a metabolite shown to decrease in processes that involve neuronal and axonal loss. Both children presented with normal volumetric MRI, MRS revealed a significant alteration in brain metabolites, with a reduction in *N*-acetylaspartate:creatine ratio for both gray an white matter compared to the subject's cousin. Trope et al. (2001) performed identical MRI and MRS studies on a sample of 16 subjects with a history of elevated blood lead levels before five years of age (23 to 65 μg/dL). Average age at time of evaluation was 8 years. These subjects were compared to age-matched controls

composed of siblings or cousins. Control subjects had blood lead levels that never exceeded 10 µg/dL. Although all of the participants had normal MRI examinations, the lead-exposed subjects exhibited a significant reduction in *N*-acetylaspartate:creatine and phosphocreatine ratios in frontal gray matter compared to controls.

Meng et al. (2005) performed MRI and MRS studies on children with blood lead concentrations ≥27 µg/dL (n = 6) and age- and gender-matched controls with blood lead concentrations <10 µg/dL (n = 6). The average age at time of evaluation was approximately 11 years. Subjects came from the Anhui province in China. Lead-exposed children had an average blood concentration of 37.7 µg/dL (SD 5.7) while controls averaged 5.4 µg/dL (SD 1.5). MRS was used to measure *N*-acetylaspartate, choline-containing compounds, and total creatine in the frontal lobes and hippocampus in cases and controls. All children presented with normal MRI with no evidence of structural abnormalities. However, peak values of *N*-acetylaspartate, choline, and creatine in all four brain regions were reduced in lead-exposed children relative to controls. The authors concluded that the reduced brain *N*-acetylaspartate levels they observed in cases may be related to decreased neuronal density or neuronal loss. Furthermore, reduced choline signal may indicate decreased cell membrane turnover or myelin alterations that can lead to central nervous system hypertrophy, while lower creatine may indicate reduced neuronal cell viability.

Using functional MRI (fMRI), Cecil et al. (2005) examined the influence of childhood lead exposure on language function in a subsample of 48 young adults from the Cincinnati Lead Study. At age 20-23 years, subjects performed an integrated verb generation/finger tapping paradigm. Higher childhood average blood lead levels were significantly associated with reduced activation in Broca's area, a recognized region of speech production in the left hemisphere. This association remained statistically significant after adjustment for the subject's latest IQ assessment. Higher childhood blood lead levels also were associated with increased activation in the right temporal lobe, the homologue of Wernicke's area (an area associated with speech production) in the left hemisphere. The results of this study suggest elevated childhood lead exposure strongly influences neural substrates of semantic language function on normal language areas with concomitant recruitment of contra-lateral regions resulting in a striking, dose-dependent atypical organization of language function.

6.3.2.8 Gene-Environment Interactions in the Expression of Lead-Associated Neurodevelopmental Deficits

The discussion of gene-environment interactions with respect to lead exposure encompasses differential susceptibilities with respect to race, gender, and genetic polymorphisms associated with lead metabolism, and neurotransmitter metabolism and function. While the differential effects of lead on neurodevelopment have been studied to some extent with respect to race and gender, very little work has been accomplished with respect to specific genetic polymorphisms.

In the U.S., African-American children are at increased risk for having an elevated blood lead level compared with white children. For example, in the last two NHANES surveys, African-American children were found to have significantly higher blood lead levels than whites, even after adjusting for urban residential status and family income (Brody et al., 1994; Mahaffey et al., 1982). However, reliable differences with respect to lead's effects on neurodevelopmental morbidity as a function of race have not been reported with consistency.

Most surveys find that boys have higher blood lead levels than girls. The data are less clear with respect to gender-related differences in lead-associated neurodevelopmental morbidities. At various assessments from birth to adolescence, a greater male vulnerability has been noted in the Cincinnati Lead Study (e.g., Dietrich et al., 1987b; Ris et al., 2004). Data from a cross-sectional study in England showed that the lead-IQ deficit association was more pronounced in boys at 6 years of age (Pocock et al., 1987). However, in a study of 764 children in Taiwan, it was found that the relationship between lead exposure and IQ scores was substantially stronger in girls (Rabinowitz et al., 1991). In the Port Pirie cohort study, lead effects on cognition were significantly stronger in girls at ages 2, 4, 7, and 11-13 years (Baghurst et al., 1992; McMichael et al., 1992; Tong et al., 2000).

At least two genetic polymorphisms have been identified that can influence the absorption, retention and toxicokinetics of lead in humans (Onalaja and Claudio, 2000). The ALAD gene has been the most studied but, as yet, the consequences of the different alleles for susceptibility to the neurodevelopmental consequences of lead exposure are unclear. Individuals with the ALAD12 or ALAD22 polymorphism tend to have higher blood lead levels than those with ALAD11. ALAD2 could increase vulnerability by raising blood lead levels or decrease it by maintaining lead in a sequestered state in the bloodstream. Only one pediatric study has

examined this directly. Bellinger et al. (1994) found that subjects with the ALAD2 polymorphism tended to have lower dentin levels than those with ALAD1. This is consistent with the concept that increased affinity of the ALAD2 polymorphism inhibits entry of lead from the blood stream into other tissues. After adjustment for exposure level, Bellinger et al. found that adolescents with the ALAD2 polymorphism performed better in the areas of attention and executive functioning assessed in their study when compared to subjects with the ALAD1 polymorphism. However, as there were only 5 subjects with the ALAD2 form, meaningful statistical comparisons could not be made.

The other gene that has been studied is the vitamin D receptor or VDR gene. This gene is involved in calcium absorption through the gut. Research on lead workers has shown that variant VDR alleles modify lead concentrations in bone, and the rate of resorption and excretion of lead over time (Schwartz et al., 2000c). Haynes et al. (2003) examined the relationship between the VDR Fok1 polymorphism and blood lead concentrations in 275 children enrolled in the Rochester Longitudinal Study. It was hypothesized that children homozygous for the *F* allele—a marker for increased calcium absorption—would have higher blood lead concentrations than heterozygotes and children homozygous for the *f* allele, after adjusting for environmental sources of lead (floor dust lead). A statistically significant interaction was found between floor dust lead loading and VDR-*Fok*1 genotypes on blood lead concentration, with the *FF* genotypes having the highest adjusted mean blood lead concentrations at 2 years of age. Consistent with other reports, Haynes et al. (2003) also found that African American children were significantly more likely to have the VDR-*FF* than were non-African American children. The ability of African American children to have increased calcium absorption may partially explain the higher blood lead concentrations observed in African American children. Unfortunately, there have been no studies to indicate which, if any, of the VDR polymorphisms are associated with increased vulnerability to the neurodevelopmental toxicity of lead.

6.3.2.9 Reversibility of Lead-related Neurodevelopmental Deficits Associated with Prenatal and Postnatal Exposure

The apparent persistence of the neurodevelopmental effects of lead observed into later childhood and adolescence has resulted in a widely held view that the damage to the central nervous system and resulting deficits in neurobehavior are irreversible. The ramifications of the

effects of lead on neurodevelopment depend not only on the extent of the initially observable effects in early childhood, but also on their enduring consequences for cognition, attainment, and behavior over the lifetime of the individual. Studies reviewed in this section are summarized in Annex Table AX6-3.10.

Since 1990, several studies attempted to eliminate or at least reduce lead-associated neurodevelopmental damage through nutritional and/or pharmacological interventions. Optimism that such interventions might be effective was raised by a New York study published in the early 1990s (Ruff et al., 1993). In an observational study, children 13 to 87 months old with blood lead levels between 25 and 55 µg/dL were given chelation with EDTA and therapeutic iron when clinically indicated. The children were then followed for 6 months. Those whose blood lead levels fell the most had improved cognitive test scores, independent of whether they had been given iron or chelation therapy. Prior to this publication, the National Institute for Environmental Health Sciences (NIEHS) was already in the process of planning a multicenter clinical trial to determine if a recently licensed oral chelating drug (dimercaptosuccinic acid or "succimer") might diminish the neurodevelopmental impact of lead in children with blood lead levels between 20 and 44 µg/dL (Rogan et al., 1998).

The Treatment of Lead-Exposed Children (TLC) study was originally designed to test the hypothesis that children with moderate blood lead levels who were given succimer would have better scores than children given placebo on a wide range of tests measuring cognition, neuropsychological functions, and behavior at 36 months of follow-up (Rogan et al., 2001). TLC enrolled 780 children from four clinical sites into a randomized, placebo-controlled, double-blind trial of up to three 26-day courses of treatment with succimer. Most children lived in deteriorating inner-city housing. Seventy-seven percent of the subjects were African American. Succimer was effective in lowering the blood lead levels of subjects on active drug during the first 6 months of the trial. However, after 1 year, differences in the blood lead levels of succimer and placebo groups had virtually disappeared. All data analyses were conducted on an intent-to-treat basis. At 36 months of follow-up, the mean IQ score on the WPPSI-R of children given active drug was 1 point lower than that of children administered placebo, and children given succimer evinced more behavioral problems as rated by the primary caregiver on the Conners Parent Rating Scale. Children given succimer scored marginally better on the Developmental Neuropsychological Assessment (NEPSY), a battery of tests designed to measure

neuropsychological deficits that can interfere with learning. However, all of these differences were statistically nonsignificant.

Although results for the first wave of follow-up for TLC were consistently negative for drug effects on cognition and behavior, they were not necessarily conclusive. Lead may affect higher-level neurocognitive processes that are inaccessible, difficult to assess, or absent in the preschool age child. In older children, scores on psychometric measures are more precise and reliable, a wider and more differentiated range of abilities can be examined, and early academic performance and social functioning outside the home environment can be evaluated. Therefore, TLC followed the cohort into the first years of elementary education to determine whether these later emerging neurodevelopmental functions were spared the effects of lead in treated children compared to placebo controls (Dietrich et al., 2004). While remaining within the limits of hypothesis driven inference, a comprehensive battery of tests were administered to TLC subjects at 7 and 7.5 years of age. These included assessments of cognition, learning, memory, global intellectual attainment, attention/executive functions, psychiatric status, behavioral and academic conduct, neurological functioning, and motor speed. However, treatment with succimer resulted in no benefit in cognitive, behavioral, neurological, and neuromotor endpoints. Indeed, children treated with succimer fared worse than children in the placebo group in several areas, including linear growth, hospitalized and outpatient injury events in the first 3 years of follow-up, and neuropsychological deficits as assessed by the Attention and Executive Functions core domain score from the NEPSY. The authors concluded that these latest follow-up data confirmed their previous finding that the TLC regimen of chelation therapy is not associated with neurodevelopmental benefits in children with blood lead levels between 20 and 44 μg/dL. Furthermore, these results emphasize the importance of taking environmental measures to prevent exposure to lead in light of the apparent irreversibility of lead-associated neurodevelopmental deficits.

In addition to pharmacological interventions, a few studies have attempted to remediate or prevent lead-associated neurodevelopmental deficits through nutritional supplementation. Recent studies attempting to reduce lead absorption through mineral hypersupplementation have been disappointing (Sargent et al., 1999). However, to date there has been only one controlled clinical trial involving lead-exposed children where central nervous system outcomes have been the focus of study. Kordas et al. (2005) and Rico et al. (2005) conducted a double-blind

nutritional supplementation trial among 602 first grade children in the city of Torreon in northern Mexico. The city is located near a metal foundry that has been a source of lead contamination in the community. The average blood lead concentration at baseline was 11.5 µg/dL (SD 6.1). About half of the children had blood lead concentrations in excess of 10 µg/dL. Subjects received 30 mg ferrous fumarate, 30 mg zinc oxide, both, or placebo daily for 6 months. In their first report, the principal outcome assessment taken at baseline and at follow-up was the parent and teacher forms of the Conners Rating Scales. There were no consistently significant treatment effects and the authors concluded that this regimen of supplementation did not result in improvements in ratings of behavior in lead-exposed children over 6 months. In addition to behavior, the authors assessed cognitive functioning with 11 tests of memory, attention, visual-spatial abilities, and learning. There were no consistent or lasting differences in cognitive performance among treatment groups confirming the earlier conclusion that nutritional supplementation alone is not effective in eliminating or reducing the impact of early lead exposure on functional neurodevelopment.

Children's blood lead levels generally decline after they peak at somewhere around 2 years of age. However, the degree of decline is a function of a number of factors including previously acquired body burden and sources of continuing exposure. Some observational studies have examined the extent to which the rate of decline in blood lead levels is associated with improvements in neurocognitive status. Tong et al. (1998) assessed the reversibility of the cognitive effects of lead in early childhood in the Port Pirie, Australia cohort study. A total of 375 children were followed to the age of 11-13 years. Average blood lead concentrations decreased from 21.2 µg/dL at 2 years to 7.9 µg/dL at 11-13 years. However, scores on standardized measures of intellectual attainment administered at 2, 4, 7, and 11-13 years of age in children whose blood lead levels declined the most were not significantly improved over those obtained by children with a more shallow decline in body burden.

Liu et al. (2002) made use of the TLC succimer trial data set (Rogan et al., 2001) to examine the question of reversibility. As reviewed above, intent-to-treat analyses revealed no benefits of chelation on neurodevelopmental indices beyond 6 months of treatment. Thus, the scores on the cognitive tests from the two treatment groups could be analyzed either within the treatment groups or as a whole. Data from 741 children were available for analyses. Mean blood lead levels in TLC subjects were 26.2 µg/dL at baseline, 20.2 µg/dL at the 6-month

follow-up, and 12.2 µg/dL at the 36-month follow-up. Mean declines in blood lead levels were 6.0 µg/dL from baseline to 6-month follow-up, 14.1 µg/dL from baseline to 36-month follow-up, and 8.0 µg/dL from 6- to 36-month follow-ups. Blood lead levels declined more quickly in the first 6 months in the succimer group than in the placebo group, but the mean blood lead levels were very similar at baseline and at the 36-month follow-up. Prior to examining changes in blood lead levels in relationship to changes in cognitive test scores, it was verified that baseline and later blood lead levels were indeed significantly associated with deficits on measures administered at specific points in the study after adjustment for socioherediatary factors surveyed in the study including maternal IQ. Unlike in the New York study by Ruff et al. (1993), Liu et al. (2002) found no overall effect of changing blood lead level on changes in cognitive test score from baseline to 6 months. However, during the follow-up from baseline to 36 months and from 6 to 36 months, falling blood lead levels were significantly associated with increased cognitive test scores, but only because of an association in the placebo group. Cognitive test scores increased by 2 points overall and 4 points in the placebo group when blood lead levels declined by 10 µg/dL from baseline to 36 months. There is a possibility that the succimer drug regimen blunted the beneficial effect. Due to the inconsistency in the results, the data do not provide strong supportive evidence that lead-induced cognitive impairments are reversible. Therefore, primary prevention and preventing additional increases in blood lead levels among children whose blood lead levels are high remain the only effective means of dealing with lead toxicity.

6.3.2.10 Periods of Enhanced Developmental Susceptibility to Central Nervous System Effects of Environmental Lead

It has been difficult to identify discrete periods of development when the fetus or child is particularly susceptible to lead's effects on neurodevelopment. When the prospective studies of lead and child development were underway, it was hoped that this methodological approach would be revealing. However, these studies observed that age strongly predicted the period of peak exposure (around 18-27 months when there is maximum hand-to-mouth activity), making it difficult to distinguish whether greater neurotoxic effects resulted from increased exposure or enhanced susceptibility at a particular age. Furthermore, children with the highest blood lead levels tended to maintain their rank order relative to their lower exposed peers throughout these

studies (e.g., Dietrich et al., 1993a; McMichael et al., 1988), limiting the degree to which investigators could identify any particular period of development as critical.

From the perspective of human neurodevelopmental biology, one could argue that the first 3 years of life should represent a particularly vulnerable period. Maximal ingestion of lead coincides with the same period of time when major events are occurring in the development of the central nervous system including some neurogenesis, rapid dendritic and axonal outgrowth, synaptogenesis, synaptic pruning, and programmed apoptosis (see Figure 6-3.4).

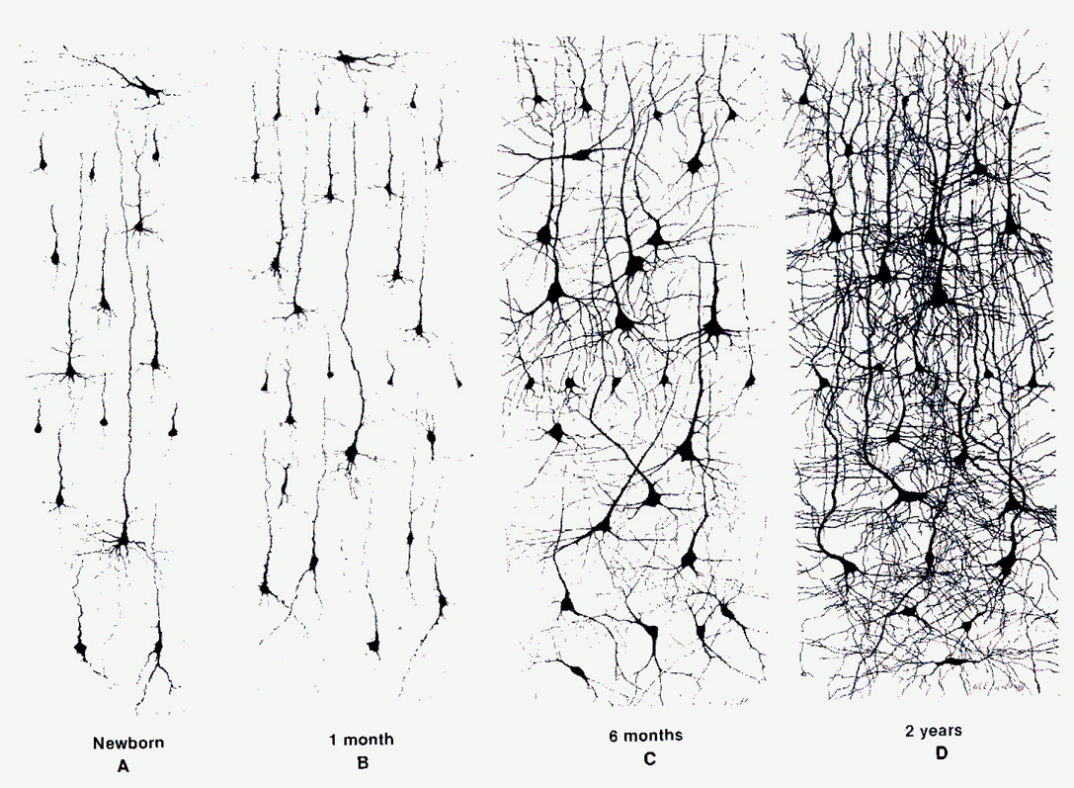

Figure 6-3.4. Golgi-stained section of human cerebral cortex taken from equivalent areas of the anterior portion of the middle frontal gyrus at different ages. Although the packing density of cortical neurons does not appear to change, there is a tremendous increase in the complexity of dendritic arborizations with increasing age with maximal density occurring between two and three years of age.

Source: Nolte (1993).

This belief that the first 3 years represents a critical window of vulnerability is evident in the lead literature (Chen et al., 2005). Two major meta-analyses of the relationships between childhood lead exposure and IQ focused primarily on the strength of the association between IQ at school age and blood lead concentrations at 2 years of age or average blood lead levels up to 3 years of age (Pocock et al, 1994; Schwartz, 1994). Neither meta-analysis considered the importance of concurrent blood lead associations in older children. The focus on these particular age groups implied that the interpretation most consistent with the overall results was that peak blood lead concentration, achieved somewhere between 1 and 3 years of age, was most likely responsible for the cognitive effects observed years later. These meta-analyses were highly influenced by findings from the Boston prospective study where blood lead concentrations at 2 years of age have been exclusively and consistently associated with lower IQ and academic achievement (Bellinger et al., 1992).

This particular interpretation of the lead literature has also influenced screening programs (which focus on 1 and 2 year olds), clinical trials that recruit children during the first 3 years of life, and current interpretation of the cross-sectional literature. For example, the report by Lanphear et al. (2000) that school-age children enrolled in the NHANES III survey displayed a significant inverse relationship between concurrent blood lead concentrations and measures of IQ and academic achievement at blood lead concentrations below 10 μg/dL was interpreted by some to reflect the effects of the children's higher blood lead concentrations when they were between 1-3 years of age.

However, it is not clear that only the period of peak blood lead concentration matters in terms of the risks for neurodevelopmental morbidity. Other prospective studies of children with both high and low lead exposures found concurrent or lifetime average blood lead levels to be more strongly associated with school age IQ and other measures of neurodevelopment (Canfield et al., 2003a; Dietrich et al., 1993a,b; Tong et al., 1996; Wasserman et al., 2000b). One study has recently attempted to address this question directly. Chen et al. (2005) sought to clarify the strength of the association between IQ and blood lead at various time points, to examine whether the cross-sectional associations observed in school age children 84-90 months of age represented residual effects from 2 years of age or "new" effects emerging among these children, and how the change in blood lead over time is related to IQ at later ages. Chen et al. (2005) used data on

780 children from the previously described TLC multicenter clinical trial (Dietrich et al., 2004; Rogan et al., 2001) to examine these relationships. Homogeneity between the two treatment groups was verified. There were no statistical differences between succimer and placebo groups in either blood lead concentrations or cognitive scores at the time points under consideration. At baseline, children were given the Bayley Scales of Infant Development. The children's full scale IQ at the 36-month follow-up was measured with the WPPSI-R. At the 60 month follow-up, IQ was assessed with the WISC-III. All neurodevelopmental outcomes were adjusted for clinical center, race, gender, language, parent's education, parent's employment, single parent family, age at blood lead concentration, and caregiver's IQ.

Figure 6-3.5 displays the mean IQ at current and subsequent ages by quartiles of blood lead measured at 2, 5, and 7 years of age. The concurrent blood lead concentration always had the strongest association with IQ. As the children aged, the relationship grew stronger. The peak blood lead concentration from baseline to 7 years of age was not associated with IQ at 7 years of age. Furthermore, in models including both prior and concurrent blood lead concentrations, concurrent blood lead was always more predictive of IQ. Adjustment for prior IQ did not fundamentally change the strength of the association with concurrent blood lead concentration. Chen et al. (2005) found a stronger relationship between IQ at 7 years of age and blood lead concentration at 7 years compared with blood lead at 2 years of age. A similar relationship was observed between IQ and blood lead at 5 years of age. The strength of the cross-sectional associations increase over time, despite lower blood lead concentrations in older children. These data support the idea that lead exposure continues to be toxic to children as they reach school age, and does not lend support to the interpretation that all of the damage is done by the time the child reaches 2 to 3 years of age. These findings also imply that cross-sectional associations observed in children, such as the study recently conducted by Lanphear et al. (2000) using data from NHANES III should not be dismissed. Chen et al. (2005) concluded that if concurrent blood lead remains important until school age for optimum cognitive development, and if 6 and 7 year olds are as or more sensitive to lead effects as 2 year olds, then the difficulties in preventing lead exposure are magnified but the potential benefits of prevention are greater.

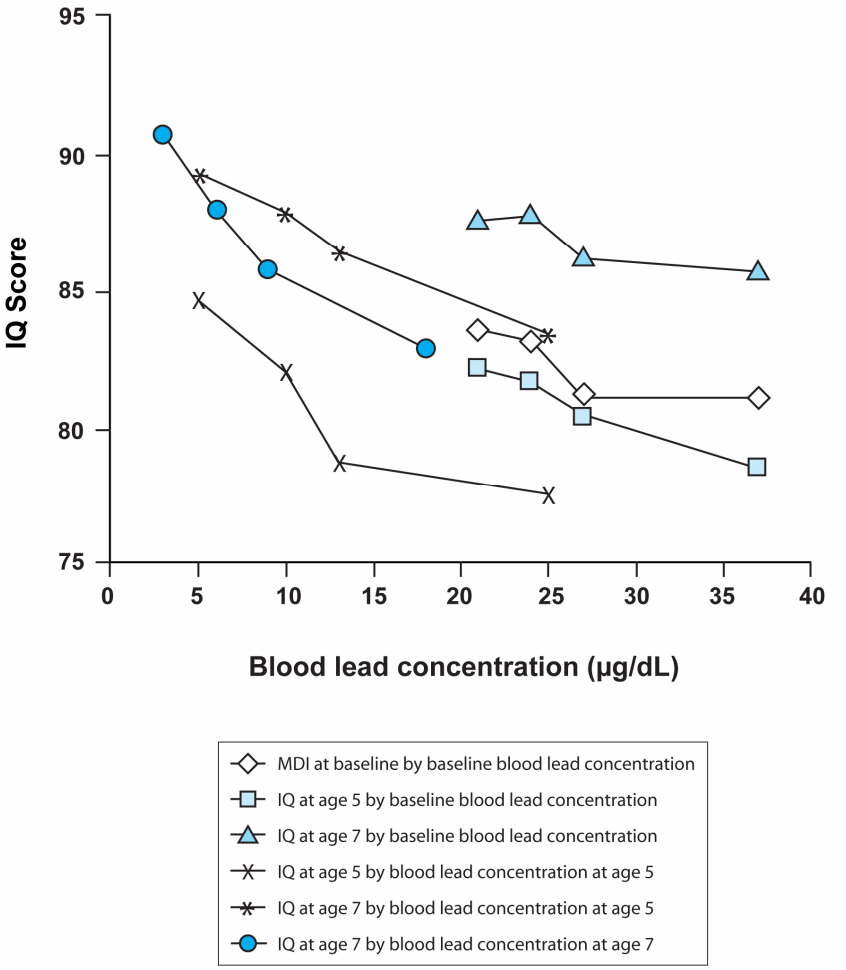

Figure 6-3.5. Full scale IQ test scores by previous or concurrent blood lead concentration. Each data point shows the mean IQ test scores of children measured at baseline or at two follow-ups, grouped by quartiles of blood lead concentration. The abscissa of each point is the middle value of each blood lead concentration category.

Source: Chen et al. (2005).

6.3.2.11 Effect of Environmental Lead Exposure on Neurodevelopment at the Lower Concentration Range

Over the last three decades, epidemiologic studies of lead and child development have demonstrated inverse associations between blood lead concentrations and children's IQ and other outcomes at successively lower levels. The 1986 Addendum and 1990 Supplement concluded that neurobehavioral effects were related to blood lead levels of 10 to 15 µg/dL and possibly

lower. In response to these data, agencies such as the U.S. Centers for Disease Control and Prevention and the World Health Organization have repeatedly lowered the definition of an elevated blood lead concentration, which now stands at 10 μg/dL (CDC, 1991; WHO, 1995). At the time when these policies were put in place, there were too few studies of children with blood lead levels consistently below 10 μg/dL on which to base an opinion as to effects at lower levels of exposure. Since the removal of lead from gasoline, the median blood lead concentration has dropped dramatically in U.S. children, permitting more studies of this nature to be done in recent years. Furthermore, the use of meta- and pooled analytic strategies has permitted investigators to get a clearer picture of effects below 10 μg/dL.

The Rochester Prospective Study (n = 172) by Canfield et al. (2003a) is illustrative. This study extended the relationship between blood lead concentrations and deficits in IQ to levels well below 10 μg/dL. Over half of the children in this study did not have a recorded blood lead concentration above 10 μg/dL. Nonlinear semiparametric smoothing revealed a covariate-adjusted decline of more than 7 points up to 10 μg/dL of childhood average blood lead and a further decline of 2 points associated with an increase from 10 to 20 μg/dL. In response to the Rochester findings, Bellinger and Needleman (2003) reanalyzed data from the Boston Prospective Study focusing on children whose blood lead levels never exceeded 10 μg/dL (n = 48). In their analyses, 10 year IQ was inversely related to blood lead levels at 24 months following adjustment for covariates. Nonparametric smoothing analyses indicated that the inverse association persisted at blood lead levels below 5 μg/dL.

Perhaps the most compelling evidence for effects below 10 μg/dL comes from an international pooled analysis of seven prospective cohort studies (n = 1,333) by Lanphear et al. (2005) described earlier. Although exposures in some cohorts were high, by pooling data from these studies a substantial number (n = 244) of children with blood lead levels that never exceeded 10 μg/dL could be included in the analyses. For the entire pooled data set, the observed decline of 6.2 points in IQ for an increase in blood lead levels from 1-10 μg/dL was comparable to the decrements for an increase in lifetime mean blood lead levels from <1-10 μg/dL observed in the Rochester Longitudinal Study (Canfield et al., 2003a). The pooled analysis of Lanphear et al. also demonstrated that deficits in IQ extended to blood lead levels <7.5 μg/dL. Therefore, recent evidence is suggestive of effects of lead on neurocognitive deficits at blood lead levels below 10 μg/dL, and possibly below 7.5 μg/dL, in children.

6.3.2.12 Selection and Validity of Neuropsychological Outcomes in Children

A fair amount of material has been written about methodologies for neurobehavioral evaluation in studies of environmental chemicals and child development (Bellinger, 2002, 2003; Dietrich et al., 2005). Much of the discussion has centered on the ability of neurobehavioral tests to detect damage to the central nervous system as a result of in utero or early postnatal exposures. In other words, the sensitivity of these tests to toxicity has been in question. The sensitivity of a neuropsychological or any other diagnostic test is defined as the proportion with the abnormality that the test classifies as abnormal (true positives). In the selection of neurodevelopmental measures in studies of lead or any other toxicant, it is clearly advantageous to include tests that have the best prognostic value. This is particularly important in the current context, because the neurobehavioral endpoints reviewed in this document are being incorporated into an assessment of risk (Bellinger, 2002). In addition, it is important to select instruments that tap into neurodevelopmental domains that have shown to be sensitive to particular environmental toxicants. As evident in this review, a large number of neuropsychological instruments, tapping a wide range of domains have proven to be sensitive to lower level lead exposure. Certain domains such as attention, executive functions, visual-spatial skills, fine-motor abilities, academic achievement (reading in particular), and externalizing behaviors appear to be affected by lead with some degree of consistency. However, the identification of behavioral phenotypes for lead has been a largely elusive goal. There are a number of plausible reasons for this. The sample's SES; level, pattern and timing of exposures; nutritional intake; general health; educational opportunities; and the particular instruments that were employed in a given study probably play an important role in between-study differences (Bellinger, 1995; Schantz, 1996). This may be one reason why the broad net provided by global, multiple domain assessments of cognition such as IQ have proven to be the most consistently sensitive across studies of various design and sample characteristics. These measures combine subscales that are representative of a broad number of underlying cognitive functions; thus, they are likely to pick up exposure-related deficits across cohorts that differ in their functional expressions of toxicity (Dietrich et al., 2005).

The validity of neuropsychological tests as indices of neurodevelopment in lead studies also is of concern. In psychometrics, there are various types of validity. But the validity lead researchers are usually most concerned about is "construct validity." If a measure has construct

validity it measures what it purports to measure. Most lead researchers utilize assessments with proven construct validity. This means that the instruments utilized by the investigator have proven that they possess concurrent and predictive "criterion" validity (i.e., it relates to other manifestations of the construct the instrument is supposed to be measuring and predicts an individual's performance in the future in specific abilities). It also means that the instrument possesses good "convergent validity." This means that the test returns similar results to other tests that purport to measure the same or related constructs. Finally, the instrument should demonstrate "discriminant validity." That is, the instrument is not measuring a construct that it is not supposed to measure, it discriminates.

Bellinger (2003) states that the general literature attests to robust observations between IQ and important measures of life success, such as grades in school, years of education, job success, social status, and income (Neisser et al., 1996; Salkever, 1995). Testing is difficult depending on examined age, especially for infants who are in a period of rapid developmental change. Also, the way an infant's cognitive function can be probed is restricted. The lack of continuity between their response modalities and ones that can be exploited as a child gets older is also a factor. Still neurobehavioral tests scores in infancy do possess strong concurrent validity.

There many potential sources of invalidity which researchers take steps to avoid. These include unreliability (an instrument that, all other things being equal, yields scores that are unrepeatable and inconsistent) and bias (e.g., due to factors such as culture, gender). Most modern standardized measures of development and cognitive attainment have taken steps to reduce these sources of invalidity and must meet certain minimum requirements such as those formulated by the American Educational Research Association, American Psychological Association, and the National Council on Measurement in Education (American Educational Research Association et al., 1999). One reason that global measures of IQ have been used so widely is because of their outstanding psychometric properties. The Wechsler series has excellent reliability and validity (Groth-Marnat, 2003). For example, the average internal consistency for the Wechsler children's scales across all age groups is 0.96. Test-retest reliability is similarly very high. The underlying factor structure of these scales has also been strongly confirmed. The validity of so-called experimental measures of learning and cognition is sometimes less certain.

All measurement procedures have the potential for error, so the goal of the researcher is to minimize it. In elementary psychometric theory, any observed test score is made of the "true" score plus measurement error. It is assumed that measurement errors are essentially random (the child's true score may not be reflected in the observed score because of errors of administration, inconsistency of administration across examiners, the child's health, or aspects of the testing environment that are not conducive to performance). This does not mean that lead researchers cannot take pains to reduce these sources of error. In fact, most modern lead researchers do minimize measurement error through attention to training, establishing inter-examiner reliability, attention to child factors, site factors, and vigilant monitoring of examiner performance throughout the course of a study (Dietrich et al., 2005).

6.3.2.13 Confounding, Causal Inference, and Effect Modification of the Neurotoxic Effect of Lead in Children

The major challenge to observational studies of lead's impact on parameters of child development has been the assessment and control for confounding factors. By definition, a confounder is associated with both the exposure and the outcome, thus has the potential to influence the association between the exposure and the outcome. Confounding by various factors can be controlled for in the design phase of the study or in the analytical phase. In the realm of lead research, there are a wide range of potential confounders, the foremost of which is SES. Socioeconomic status is measured rather crudely in most studies with such indices as the Hollingshead Four-Factor Index of Social Position that incorporates education and income of both parents. However, even these so-called blunt measures often account for a great deal of the variance in neurodevelopmental outcomes. Given the crude nature of these measures, to control for confounding by SES as well as rearing environment of the child, many recent lead studies have incorporated more direct assessments such as the HOME scale, parental intelligence, parental attitude assessments, and measures of parental substance abuse and psychopathology. Given the relatively high correlation between indices of lead exposure and social environmental factors, the consistency among studies in finding effects following adjustment for these confounding factors is remarkable. It is important to consider the enormous experimental animal evidence not compromised by the possibility of confounding in examining lead effects on health (Bellinger, 2004; Davis et al., 1990; U.S. Environmental Protection Agency, 1986a, 1990).

Another problem in the analyses of data on lead and child development is the lack of critical consideration of which potential confounder in a particular model "owns" the variance in neurodevelopmental performance. Thus, for example, in the case of social class it is assumed that if an effect of lead is reduced to nonsignificance following adjustment for some measure of socioeconomic standing, the assumption is that all of the variance belongs to the confounder. However, in some instances this could be seen as an excessively conservative interpretation and raises the specter of Type II error. Social class could be seen as either a confounder or a proxy for exposure. Lower social class in urban children is closely linked to residence in older housing in poor condition that, in turn, is associated with higher levels of environmental lead (Clark et al., 1985). If studies adjust for social class in the usual manner, the effects of the toxicant will be underestimated (Bellinger, 2004). One extreme example of overcontrol of this nature can be found in the New Zealand studies where investigators regularly "controlled" for residence in older "weatherboard" housing (e.g., Fergusson et al., 1988a,b). However, it is worth noting that even in the models including this variable lead remained a significant predictor of intellectual and academic under-attainment in the Christchurch Health Study.

Most of the important confounding factors in lead studies have been identified and efforts have been made to control them in studies conducted since the 1990 Supplement. Invocation of the poorly measured confounder as an explanation for positive findings is not substantiated in the database as a whole when evaluating the impact of lead on the health of U.S. children (Needleman, 1995). Of course, it is often the case that following adjustment for factors such as social class, parental neurocognitive function, and child rearing environment using covariates such as parental education, income, and occupation, parental IQ, and HOME scores, the lead coefficients are substantially reduced in size and statistical significance (Dietrich et al., 1991). This has sometimes led investigators to be quite cautious in interpreting their study as positive (Wasserman et al., 1997). This is a reasonable way of appraising any single study, and such extreme caution would certainly be warranted if forced to rely on a single study to confirm the lead effects hypothesis. Fortunately, a large database of high quality studies on which to base inferences regarding the relationship between lead exposure and neurodevelopment exists. In addition, lead has been extensively studied in animal models at doses that closely approximate the human situation. Experimental animal studies are not compromised by the possibility of confounding by such factors as social class and correlated environmental factors. The enormous

experimental animal literature that proves that lead at low levels causes neurobehavioral deficits and provides insights into mechanisms is to be considered when drawing causal inferences (Bellinger, 2004; Davis et al., 1990; U.S. Environmental Protection Agency, 1986a, 1990).

In addition to being a confounder, social class and related variables have been shown to be effect modifiers in many studies of lead and child development (Bellinger, 2000; Tong et al., 2000). Effect modification occurs when the magnitude of an association between an exposure (lead) and an outcome (neurobehavior) varies across strata of some other factor (Last, 2001). The disadvantages that accompany poor education and underemployment have been found to exacerbate the effects of lead when carefully examined (Bellinger et al., 1989). Indeed, evaluating potential effect modifiers should be considered an important part of an overall data analytic plan.

6.3.3 Summary of the Epidemiologic Evidence for the Neurotoxic Effects of Lead in Children

Effects of lead on neurobehavior have been detected with remarkable consistency across numerous studies of various designs, populations studied, and developmental assessment protocols. The negative impact of lead on IQ and other neurobehavioral outcomes persist in most recent studies following adjustment for numerous confounding factors including social class, quality of caregiving, and parental intelligence.

Three meta-analyses and one international pooled analysis of seven prospective studies have confirmed that exposure to lead at low dose has an effect on the intellectual attainment of preschool and school age children. Recent analyses examining the association of lead with intellectual attainment and academic performance in children with low lead exposures have observed effects at blood lead concentrations below 10 μg/dL. The pooled analysis by Lanphear et al. (2005) observed a decline of 6.2 points (95% CI: 3.8, 8.6) in full scale IQ for an increase in concurrent blood lead levels from 1 to 10 μg/dL.

The effects of lead on behavior and mood of children has been an area of recent research. These studies have demonstrated that the impact of lead may extend into increased risk for antisocial and delinquent behavior. This may be a consequence of attentional problems and academic underachievement among children who have suffered higher exposures to lead during

their formative years. Several studies that have used methods of MRI and MRS to assess direct measures of brain damage also are suggesting evidence of harm due to lead exposure.

Attempts to eliminate or limit lead-associated neurodevelopmental morbidities with pharmacological or nutritional intervention strategies have been shown to be ineffective, further emphasizing the importance of taking environmental measures to reduce and possibly prevent exposure to lead in children.

6.3.4 Summary of Key Findings on the Neurotoxic Effects of Lead in Adults from the 1986 Lead AQCD

Lead intoxication in adults occurred primarily in occupational settings with historically high exposure levels. In more recent times, occupational lead exposure has been reduced to much lower levels and is often associated with no symptoms. The symptom constellation associated with high levels of lead exposure include impaired memory and attention span, irritability, headache, muscular tremors, and hallucinations (Cantarow and Trumper, 1944) that may progress to signs of frank encephalopathy (Smith et al., 1938). Symptoms of lead intoxication begin with blood lead >40 μg/dL (Baker et al., 1979) accompanied by poorer performance on cognitive and visuomotor tasks, reaction time, verbal learning, and reasoning ability that reflect involvement of both the central nervous system and the peripheral nervous system (Arnvig et al., 1980; Campara et al., 1984; Grandjean et al., 1978; Haenninen et al., 1978, 1979; Hogstedt et al., 1983; Mantere et al., 1982; Valciukas et al., 1978; Zimmermann-Tansella et al., 1983). Impaired occulomotor function, measured by saccade accuracy and velocity, depended upon the age group of the lead-exposed worker (Baloh et al., 1979; Glickman et al., 1984; Spivey et al., 1980).

With regard to peripheral nerve function as measured by nerve conduction studies, the 28 studies reviewed by the U.S. EPA in the 1986 Lead AQCD found no consistent single nerve involved but, overall, the exposed group had slower conduction velocity at blood lead concentrations as low as 30 μg/dL.

Studies reviewed in 1986 found that amyotrophic lateral sclerosis (ALS) was inconsistently associated with elevated lead levels in the nervous system. Chelation for 1 year did not did not alter elevated lead levels in the tissue of patients with motor neuron disease.

6.3.5 Neurotoxic Effects of Lead in Adults

6.3.5.1 Overview of Cognitive and Psychomotor Tests Associated with Adult Lead Exposure

Examination of lead effects on neurobehavioral performance in adults differs from that in children, since the neurobehavioral tests in adults focus on loss of abilities previously present rather than the lack of attainment of those abilities. Also, there is contribution of cognitive reserve acquired by years of education, self-education, on-the-job training, avocational, and non-avocational activities that increases the ability to compensate for the effects of lead exposure on learning new information. Medical conditions requiring medications, head trauma, and other neuropsychiatric conditions that impact nervous system performance have increased in prevalence in the adult population. These factors may increase the impact of lead exposure or be mistaken for the effects of lead and, therefore, must be handled in the analysis.

As alterations in mood may be associated with lead exposure, many neurobehavioral batteries use self-administered questionnaires to screen for mood. The Center for Epidemiologic Studies Depression Scale (CES-D) screens for depression. The Profile of Mood State (POMS) screens for six subscales, namely anger, confusion, depression, fatigue, anxiety/tension, and vigor. The six mood scales of the POMS were originally validated in a clinical psychiatric population; thus, the factor structure needed to be validated in an occupational population. Factor analysis of the POMS in lead smelter workers found only two relevant factors: (1) "general distress," composed of the subscales anger, confusion, depression, fatigue, and tension; and (2) "psychological adjustment," which contained vigor (Lindgren et al, 1999). This brings into question the use of the six scales as separate outcome variables in the study of lead exposure.

Neurobehavioral tests commonly used to demonstrate the effects of lead are listed below (for a more complete description, see Lezak, 1995). Mini-Mental-State Examination (MMSE), a screening tool for cognitive impairment, is a compilation of many cognitive domains including orientation to time and place, registration, and recall of three words, attention, language, and visual construction with a total possible score of 30 (Folstein et al, 1975). MMSE is sensitive to age and education. In 194 healthy subjects aged 40 to 89 years with 7-21 years of education, only 1% of the subjects obtained an MMSE score of 24/30 and none below (Bleecker et al., 1988). MMSE errors are sensitive to age effects including delayed recall, spelling "WORLD"

backwards and repetition of "no ifs, ands, or buts." With lead exposure, examination of errors is important to compare with age-related changes and to determine the biological plausibility of the effects of exposure especially when performing repeated measures of the test.

Neurobehavioral batteries should always include a benchmark test such as Vocabulary or a reading test such as the Wide Range Achievement Testing for Reading (WRAT) or the North American Reading Test (NART) that are considered to be resistant to neurotoxic exposure. Results from these tests should be adjusted for in the analysis. In blue-collar workers, this may be a better measure of educational achievement than years of education (Bleecker et al., 2002).

Neuropsychological batteries screening for the effects of lead usually include the following domains (Lezak, 1995): attention/concentration (Digit Span); conceptual and executive functioning (Stroop, Trails B); visuoperceptive/visuoconstructive (Block Design); visuomotoric (Reaction Time, Pegboard Test, Digit Symbol Substitution, Trails A); verbal memory (Rey Auditory Verbal Learning Test, Logical Memory, Paired Associated Learning); and nonverbal memory (Rey-Osterreith Complex Figure, Benton Visual Retention). When analyzing the association of lead exposure and test performance, adjusting for potential confounders is critical. Potential confounders are namely age, education (preferably a measure of verbal intelligence), depressive symptoms, alcohol use, and smoking. In some cases, age (Bleecker et al., 1997a) and education (Bleecker et al., 2002) may serve as effect modifiers. The association of lead and poorer neurobehavioral outcome has been found to be present only in older workers or those with less education.

6.3.5.2 Neurobehavioral Effects Associated with Environmental Lead Exposure

Exposure to chronic low levels of environmental lead and its association with effects on the nervous system were examined in several populations originally followed to study conditions associated with aging: the VA Normative Aging Study (NAS) (Payton et al., 1998; Weiskopf et al., 2004; Wright et al., 2003); the Study of Osteoporotic Fractures (Muldoon et al., 1996); and the Kungsholmen Project on aging and dementia (Nordberg et al., 2000). Studies reviewed in this section are summarized in Annex Table AX6-3.11.

The VA Normative Aging Study (NAS) is a multidisciplinary longitudinal investigation of the aging process established in 1963 and conducted at the VA Outpatient Clinic in Boston, MA. The NAS cohort cannot be considered to be exclusively representing the general

population, as bone lead measurements are higher than expected for only environmental exposure and, thus, suggest the possibility of other sources such as past occupational exposure, diet, and drinking water (Elmarsafawy et al., 2002, Vijayalakshmi et al., 1999).

The relationship of bone lead and blood lead to psychiatric symptoms in NAS (Rhodes et al., 2003) found mood symptoms for anxiety and depression potentially associated with bone lead levels. Education also was inversely related to bone lead; however, high school graduates had significantly higher general stress that may be related to SES and not lead exposure.

Neuropsychological testing in NAS found response speed sensitive to low levels of lead but it was not a consistent finding in all tests measuring the same domain upon examination of 141 healthy men with a mean age of 67 years, education 14 years. The mean blood lead level was 6 μg/dL, patella bone lead was 32 μg/g bone mineral, and tibia bone lead was 23 μg/g bone mineral (Payton et al., 1998). Vocabulary, a measure of verbal intelligence and predictor of neurobehavioral performance, was used as an outcome variable instead of being adjusted for as a potential confounder. Education was negatively correlated with bone lead and blood lead, suggesting other factors bedsides lead exposure may have contributed to neuropsychological performance. The handling of multiple comparisons was not addressed.

Another analysis of the NAS (Wright et al., 2003) examined 736 men, mean age 68 years with education level of 54% high school or less. The mean blood lead was 5 μg/dL, and mean patellar and tibia lead levels were 30 and 22 μg/g bone mineral, respectively. The subjects had a mean MMSE score of 27. Relation of MMSE scores <24 (n = 41) and blood lead by logistic regression estimated an odds ratio of 1.21 (95% CI: 1.07, 1.36). For patella lead and tibia lead, odds ratios of 1.21 (95% CI: 1.00, 1.03) and 1.02 (95% CI: 1.00, 1.04), respectively, were observed. Risk of MMSE <24 (6% of the present population versus 1% of previously described healthy aging study) when comparing the lowest and highest quartiles was 2.1 (95% CI: 1.1, 4.1) for patella lead, 2.2 (95% CI: 1.1, 3.8) for tibia lead, and 3.4 (95% CI: 1.6, 7.2) for blood lead. Interaction of age with patella lead and blood lead in predicting MMSE found steeper decreases in MMSE scores relative to age in the higher quartiles of patella lead and blood lead. Types of errors on the MMSE were not included. It was not addressed how medical conditions and medications that occurred over the duration of the study and could potentially affect cognitive performance were handled. If the community dwelling population in NAS (Wright et al., 2003) had older individuals with chronic medical conditions and less education

(213 subjects had an education less than high school) living in areas with higher past lead pollution, the confounding may be impossible to sort out.

Weisskopf et al. (2004) expanded the MMSE study in NAS by examining 466 men, mean age 70 years who had completed the MMSE twice with an interval of approximately 3.5 years. Mean blood lead was 4 μg/dL, and mean patella and tibia bone lead were 23 and 19 μg/g bone mineral, respectively. Baseline mean MMSE score was 27 and mean change in MMSE score over 3.5 years was 0.3 points. Even though MMSE change was significantly associated with bone lead, a change in MMSE score by a fraction of a point does not constitute a meaningful change of cognitive performance. To address the biological plausibility of change in the MMSE over 3.5 years, errors by functional domain need to be identified to rule out the possibility of random change with repeat performance.

Muldoon et al. (1996) studied participants in the Study of Osteoporotic Fractures for an association of nonoccupational lead exposure and cognitive function. The Study of Osteoporotic Fractures began in 1986 and included women over age 65 years living in four different communities – Baltimore, MD; Portland, OR; Minneapolis, MN; and the Monongahela Valley outside of Pittsburgh, PA. A sample of 325 women from rural sites with a mean age of 71 years (mean blood lead 4.5 μg/dL) and 205 women from urban sites with a mean age of 69 years (mean blood lead 5.4 μg/dL) were examined. The urban group was more educated and had higher use of cigarettes and alcohol. Performance examined by blood lead groups adjusting for age, education, smoking, and alcohol use found no significant differences in the urban group. However, in the rural group, individuals with blood lead >7 μg/dL had significantly poorer performance when compared to those with blood lead <4 μg/dL for Trails B, Digit Symbol, and Reaction Time. Response time across blood lead groups increased for the rural group and decreased or remained the same for the urban group. Mean MMSE for the whole population was 25, with poorer performance in the rural group—thus, suggesting an increased prevalence of clinical cognitive disorders of another etiology. Even though the neuropsychological battery was simple, 9 participants were unable to perform some of the tests including 3 on the MMSE. Such severe impairments were not found with higher occupational exposures, which raises the question as to whether other factors not measured accounted for these differences attributed to blood lead.

In the Kungsholmen Project on aging and dementia in Stockholm, Sweden, no relationship was found between blood lead and MMSE (Nordberg et al., 2000). The study population included 762 participants with a mean age of 88 years. The mean blood lead in this group was 3.7 μg/dL and the mean MMSE was 25. In contrast to the other populations examined, this study cohort was more homogenous, comprised entirely of elderly Swedes. Their likelihood of prior exposure to elevated lead levels was low.

Overall, these studies of environmental lead exposure in adults are difficult to interpret, as many competing risk factors for neurobehavioral performance in the elderly were not considered. Also, bone lead levels were higher than expected from environmental exposure suggesting unrecognized previous occupational exposure. The association of bone lead with neurobehavioral performance was unusual, in as much as it was not demonstrated in studies of occupational exposure (reviewed below). At this time, these studies do not demonstrate that the aging nervous system is at increased risk for poorer neurobehavioral performance related to environmental lead exposure as reported in children.

6.3.5.3 Neurological Symptoms Associated with Occupational Lead Exposure

Studies reviewed in this section are summarized in Annex Table AX6-3.12. Several occupational studies found blood lead levels of 29-43 μg/dL associated with POMS subscales (Hänninen et al., 1998; Maizlish et al., 1995; Niu et al., 2000). However, other studies with blood lead levels of 27-38 μg/dL found no relationship with POMS (Chia et al., 1997; Lucchini et al., 2000; Osterberg et al., 1997; Stollery et al., 1989). A screen for depression, CES-D, was administered to 803 lead-exposed Korean workers. CES-D was significant associated with tibia lead (mean 37 μg/g bone mineral), but not with blood lead (mean 32 μg/dL), after adjusting for covariates (Schwartz et al., 2001a).

Dimercaptosuccinic acid (DMSA)-chelatable lead reflects the mobilizable fraction of lead in the soft tissue. Korean lead-exposed workers (n = 95) with DMSA-chelatable lead (mean 289 μg) above the median of 261 μg were 6.2 times more likely to have tingling or numbness in their extremities, 3.3 times more likely to experience muscle pain, and 3.2 times more likely to feel irritable (Lee et al., 2000). The workers with higher chelatable lead were 7.8 times more likely to experience neuromuscular symptoms compared to workers with lower chelatable lead. Blood zinc protoporphyrin predicted weakness of ankle and wrist and fatigue while delta-

aminolevulinic acid (ALAD) in urine (mean 3 mg/L) predicted inability to sleep; however, blood lead (mean 45 μg/dL) was not significantly associated with any symptoms.

In some studies, difficulty concentrating, irritability, fatigue, muscle pain, and joint pain were more likely in workers with a mean blood lead of 43 μg/dL (Maizlish et al., 1995) and 27 μg/dL (Lucchini et al., 2000), whereas other studies with mean blood lead >30 μg/dL found no association with symptoms (Chia et al., 1997; Osterberg et al., 1997). Lucchini et al. (2000) provided an estimated threshold of blood lead 12 μg/dL for significant increase of neurological symptoms.

In summary, even though one study suggested a threshold for neurological symptoms at a blood lead of 12 μg/dL (Lucchini et al., 2000), other studies with blood lead >30 μg/dL found no association with lead-related symptoms. The study by Lee et al. (2000) observed that higher levels of DMSA-chelatable lead was associated with irritability, tingling or numbness in their extremities, muscle pain, and neuromuscular symptoms.

6.3.5.4 Neurobehavioral Effects Associated with Occupational Lead Exposure

Studies reviewed in this section are summarized in Annex Table AX6-3.13. Discriminate analysis of neurobehavioral performance found the group of tests that best differentiates lead-exposed workers (mean blood lead 49 μg/dL) from nonexposed workers were Simple Reaction Time (SRT), Digit Symbol (WAIS), and Trail Making Test (Part A) (Boey et al., 1988). Using a similar battery with 44 lead-exposed workers, mean blood lead 29 μg/dL, performance was significantly associated with blood lead for SRT, digit symbol and pursuit aiming (Niu et al., 2000).

Seventy workers grouped by blood lead (<20, 21-40, and 41-80 μg/dL) were examined on three occasions each separated by 4 months. Performance on reaction time was stable except in the high lead group where decision time was slowed more than movement time along with concentration difficulties that remained consistently across testing sessions. Memory testing did not improve with repetition in the high lead group (Stollery et al., 1991). Decision gaps as opposed to movement gaps were selectively affected by lead exposure in this population (Stollery, 1996).

A review of occupational lead exposure in 1995 (Balbus-Kornfeld et al., 1995) concluded that the association of cumulative lead exposure or body burden of lead and neurobehavioral

performance in adults was inadequately covered in the literature. Studies have addressed these deficiencies with the use of a working lifetime integrated blood index and bone lead concentrations. Even though exposure assessment has improved, there is variability based upon differences in past exposure versus present exposure, duration of exposure, frequency of monitoring for blood lead, lead exposure from other occupational sources and nonoccupational activities. Measurement of bone lead addresses some of these problems but the relationship of bone lead concentration and lead levels in the brain or peripheral nervous system is inconsistent.

Subsequent studies used measures of cumulative lead exposure, namely lifetime integrated blood index, weighted average blood lead, and bone lead. More consistent associations occurred with the lifetime integrated blood index and weighted average blood lead for visuomotor/visuoperceptive tasks of Pegboard, Pursuit Aiming, Digit Symbol, Trails, and Block Design (Bleecker et al., 1997a; Chia et al., 1997; Hänninen et al., 1998; Lindgren et al., 1996; Schwartz et al., 2005) while others found no association with these lead exposure measures (Lucchini et al., 2000; Osterberg et al.,1997; Schwartz et al., 2001a). Age served as an effect modifier for the association of the lifetime integrated blood index with pegboard (Bleecker et al., 1997a).

One difficulty with cumulative lead dose is the inability to separate the effect of past high exposure from a lower proximate exposure. To address this issue, workers with similar past high exposure were grouped by those with proximate exposure above blood lead of 40 μg/dL and those with proximate exposure below blood lead of 40 μg/dL and were compared on performance of verbal memory (Lindgren et al., 2003). Use of regression analyses found pattern group contributed significantly to the explanation of variance in verbal memory after adjusting for current blood lead and lifetime integrated blood index measures. The relationship between past high exposure and verbal memory no longer existed in the group that maintained proximate blood lead below 40 μg/dL, suggesting the possibility of reversibility of the effects of lead in adults.

The first study to report the effects of cumulative lead exposure on the nervous system examined 467 Canadian lead smelter workers, with a mean of 18 years of employment (Lindgren et al., 1996). Their mean blood lead level was 28 μg/dL, the weighted average blood lead level was 40 μg/dL, and the lifetime integrated blood index was 765 μg·year/dL. Fourteen neuropsychological variables were examined by MANCOVA using exposure groups of high,

medium, and low. The analysis was adjusted for age, education, years employed, CES-D, and alcohol use. Exposure groups categorized using lifetime integrated blood index differed significantly on digit symbol, logical memory, Purdue dominant hand, and Trails A and B. No concentration-response relationship between blood lead and neuropsychological performance was found. From this smelter population, 256 workers currently employed had a median MMSE score of 29 (range 19-30). A concentration-response relationship between lifetime integrated blood index and MMSE was found only in the 78 workers with a WRAT-R reading grade level less than 6. The absence of a concentration-response relationship in workers with higher reading grade levels and the same lifetime integrated blood index dose was attributed to increased cognitive reserve (Bleecker et al., 2002). An in-depth examination of verbal learning and memory in this same population found no association with blood lead. However, increasing lifetime integrated blood index or weighted average blood lead was associated with poorer storage and retrieval of previously learned material. Alterations in the ability to organize materials in long-term memory interfered with retrieval efficiency (Bleecker et al., 2005a). The one test sensitive to blood lead in this smelter population was simple reaction time that had a curvilinear relationship with slowing beginning at a blood lead of approximately 30 μg/dL (Bleecker et al., 1997b).

Fifty-four lead battery workers were stratified by those whose blood lead never exceeded 50 μg/dL (n = 26) and those who had higher exposure in the past (n = 28) to examine the neuropsychological effects of current low level blood lead versus those of higher blood lead in the past (Hänninen et al., 1998). Partial correlations controlling for age, sex, and education in the low group found block design, digit symbol, digit span, similarities, Santa Ana 1, and memory for design associated with recent measures of exposure and embedded figures with maximum blood lead (mean maximum blood lead 40 μg/dL). Embedded figures, digit symbol, block design, and associative learning were associated with the lifetime integrated blood index (mean 823 μg·year/dL) and maximum blood lead (mean 69 μg/dL) in the high blood lead group. There was essentially no association with bone lead in either group. A concentration-response relationship existed for digit symbol, embedded figures, and memory for design. Overall past high exposure with blood lead levels >50 μg/dL had the greatest effect on tests requiring the encoding of complex visually presented stimuli. The authors concluded that the effect of lead on brain function was better reflected by history of blood lead than content of lead in bone.

However, some studies that included measures of cumulative lead and current lead exposures found the strongest association with current blood lead. Schwartz et al. (2001a) examined the associations of blood lead, DMSA-chelatable lead, and tibia lead with neurobehavioral tests in 803 Korean lead-exposed workers from a variety of industries and 135 controls. In lead-exposed workers, the mean blood lead level was 32 μg/dL, DMSA-chelatable lead level was 186 μg, and bone lead levels was 37 μg/g bone mineral, compared to controls with a mean blood lead level of 5 μg/dL and bone lead level of 6 μg/g bone mineral. Compared to controls, lead-exposed workers performed significantly worse on SRT, Digit Span, Benton Visual Retention, Colored Progressive Matrices, Digit Symbol, and Purdue Pegboard after controlling for age, gender, and education. The association of DMSA-chelatable lead with test performance became nonsignificant after the addition of blood lead in the model. Bone lead was not associated with neurobehavioral performance. Blood lead was the best predictor for significant decrements in neurobehavioral performance on Trails B, Purdue Pegboard (4 measures) and Pursuit Aiming (2 measures). The effect of a 5 μg/dL increase in blood lead was equivalent to an increase of 1.05 years in age. Use of LOWESS functions suggested a threshold at blood lead 18 μg/dL after which there is a decline of performance in Purdue Pegboard (assembly) and Trails B.

From the above cohort of Korean lead workers, 212 consecutively enrolled workers were examined for protein kinase C (PKC) activity and the relations between blood lead and neurobehavioral performance (Hwang et al., 2002). Blood lead range from 5 to 69 μg/dL was associated significantly with decrements in Trails B, SRT, and Purdue Pegboard (3 measures). PKC activity, as measured by back-phosphorylation of erythrocyte membrane proteins, was not associated with neurobehavioral test scores. Addition of the interaction term of blood lead with PKC activity dichotomized at the median found significant effect modification with the association of higher blood lead and poorer neurobehavioral performance occurring only among workers with lower PKC activity that corresponds to higher in vivo PKC activity. The authors suggested that PKC activity might identify a subpopulation at increased risk of neurobehavioral effects of lead.

Occupational lead exposure and longitudinal decline in neurobehavioral performance was examined in 576 current and former Korean lead workers who completed testing at three visits at approximately yearly intervals (Schwartz et al., 2005). At baseline, the mean blood lead was

31 µg/dL and the mean tibia lead was 38 µg/g bone mineral. Blood lead from baseline correlated with those from visit 2 and 3 and baseline tibial lead correlated with that measured at visit 2. Cross-sectional associations of blood lead or short-term change occurred with Trails A and B, Digit Symbol, Purdue Pegboard (4 measures), and Pursuit Aiming after adjusting for potential confounders. However, longitudinal blood lead was only associated with poorer performance on Purdue Pegboard (4 measures). Historical tibial bone lead was associated with digit symbol and Purdue Pegboard (dominant hand). Magnitude of lead associations was expressed as the number of years of increased age at baseline that was equivalent to an increase of lead from the 25th to 75th percentile. The effect of cross-sectional blood lead at baseline was equivalent to 3.8 years of age, 0.9 years of age for historical tibial lead, and 4.8 years of age for longitudinal blood lead.

In summary, performances on visuomotor and verbal memory tasks are consistently associated with occupational lead exposure. In several studies, cumulative blood lead index was found to be a strong predictor of neurobehavioral performance. Lead concentrations in bone were a weaker predictor of lead effects on brain function.

6.3.5.5 Neurophysiological Function and Occupational Lead Exposure

A meta-analysis including 32 nerve conduction studies with occupational lead exposure found blood lead to be a weak predictor of peripheral nerve impairment (Davis and Svendsgaard, 1990). Nerve conduction velocities were reduced in lead-exposed subjects, with the greatest sensitivity observed in the median motor nerve. Decreasing effect sizes were observed with increasing duration of exposure. Meta-analyses of neurobehavioral effects in adults are presented in Annex Table AX6-3.14.

Studies reviewed in this section are summarized in Annex Table AX6-3.15. Nerve conduction studies of workers in a lead battery factory (Kovala et al., 1997) found sensory amplitudes of the median and sural nerves had a negative correlation with long-term exposure (lifetime integrated blood index and duration of exposure). Chia et al. (1996b) also found the strongest concentration-response relationship between median sensory conduction velocity and lifetime integrated blood index. He et al. (1988) found sensory conduction abnormalities related to blood lead levels.

Yokoyama et al. (1998) measured the distribution of conduction velocities in large myelinated fibers of the sensory median nerve twice at a year interval in 17 gunmetal workers.

In workers with a 1-year change in chelatable-lead (mobilized lead) greater than 440 μg/24 h, conduction velocities of faster fibers were decreased significantly. Measure of body burden (readily mobilized lead from soft tissue) was a stronger predictor of peripheral nerve impairment than blood lead.

A group of studies examined vibration threshold in the extremities (Chuang et al., 2000; Kovala et al., 1997; Schwartz et al., 2001a, 2005). In 60 workers exposed to lead, Kovala et al. (1997) found vibration threshold at the ankle related to the lifetime integrated blood index and duration of exposure while the finger vibration threshold was associated with current blood lead exposure. Overall historical blood lead measures were more closely associated with peripheral nerve function than bone lead in this population. By contrast, Schwartz et al. (2001a) also examined vibration thresholds and bone lead in 803 Korean workers and 135 controls and found that tibia lead (mean 37 μg/g bone mineral) but not blood lead (mean 32 μg/dL) was significantly associated with poorer vibration threshold in the dominant great toe but not the finger. In a follow-up study of 576 lead workers who completed three visits at yearly intervals, vibration threshold in the toe was associated with current blood lead (mean 31 μg/dL), longitudinal blood lead, and tibia lead (38 μg/g) after adjusting for covariates (Schwartz et al., 2005). Chuang et al. (2000) reported on vibration perception in the foot in 206 lead battery workers. There was a significant association with blood lead (mean 28 μg/dL) and weighted average blood lead (mean 32 μg/dL) with vibration perception in the foot after adjustment for covariates including the use of vibrating hand tools. A hockey stick regression analysis of foot vibration threshold versus mean blood lead concentration for the past 5 years found an inflection point around 30 μg/dL with a positive linear relation above this point, suggesting a potential threshold.

Bleecker et al. (2005b) examined peripheral nerve function in 80 smelter workers using Current Perception Threshold (CPT), a neuro-selective test that measures integrity of the large and small myelinated nerve fibers and unmyelinated nerve fibers. CPT was not associated with blood lead (mean 26 μg/dL) or bone lead (mean 40 μg/g bone mineral). CPT for large myelinated nerve fibers had a curvilinear relationship with weighted average blood lead (mean 42 μg/dL), with a threshold observed at 28 μg/dL. Results from further regression analyses suggested that even with cumulative lead exposure, intensity is more important than duration of exposure to lead with regard to the peripheral nervous system. At the highest criterion blood lead level, both large and small myelinated nerve fibers were impaired. The presence of

activated motor units, equated to ergonomic stressors by job title, enhanced the effect of lead on the peripheral nervous system.

In summary, occupational lead exposure studies consistently found peripheral sensory nerve impairment as opposed to the classic motor neuropathy described historically with high lead exposure. A possible threshold for this effect on the sensory nerves was observed at a blood lead of 28 µg/dL.

6.3.5.6 Evoked Potentials and Occupational Lead Exposure

Visual evoked potentials (VEPs) and brainstem auditory evoked potentials (BAEPs) measure speed of conduction in the visual and auditory pathway. BAEPs have discrete waveforms with wave I arising from the auditory nerve; its latency reflects peripheral transmission time. Wave III is predominantly generated from the caudal pons and wave V from the inferior colliculus. The use of interpeak latencies removes abnormalities in the auditory nerve latency from changes in brainstem transmission in the auditory pathway. Studies reviewed in this section are summarized in Annex Table AX6-3.16.

Abbate et al. (1995) performed VEPs on 300 lead-exposed men (aged 30 to 40 years) in good health with no other neurotoxic exposure. Range of blood lead levels was 17 to 60 µg/dL. Individuals were stratified into 4 groups with mean blood lead levels of 23 µg/dL (n = 39), 30 µg/dL (n = 113), 47 µg/dL (n = 89), and 56 µg/dL (n = 59). P100 latency measured for VEPs were significantly prolonged across the blood lead groups. Linear regression found the association of blood lead and P100 were significant in each group, but the relationship was not proportional. Prolonged VEP began at a blood lead levels of 17-20 µg/dL. With age limited to one decade, contribution from age was not a concern. Even though there was no comparison group, careful screening ruled out other medical and eye conditions, and other potential exposures.

BAEPs recorded in 49 lead-exposed workers and age and sex matched controls (Discalzi et al., 1992) had mean blood lead levels of 55 µg/dL and a mean weighted average blood lead level of 54 µg/dL. Interpeak latencies, I-V, I-III, and III-V were all prolonged in the lead-exposed workers. In those workers with weighted average blood lead >50 µg/dL, I-V latency was longer. Discalzi et al. (1993) reported identical results in a subsequent publication of 22 battery storage workers with a mean blood lead of 47 µg/dL and a mean weighted average

blood lead of 48 μg/dL. Holstein et al. (1986) examined 20 adults accidentally exposed to lead through food until 1 year prior to the study. On the day of examination, the mean blood lead level was 31 μg/dL, while the mean weighted average blood lead was 43 μg/dL. Latencies I, III, and I-III interpeak intervals were longer in the exposed group with a concentration-response relationship observed for the weighted average blood lead and I-III interpeak interval.

BAEPs were performed in 359 currently-employed smelter workers with a mean of 17 years of employment. The mean blood lead levels was 28 μg/dL (SD 8.4), mean working-lifetime weighted average blood lead was μg/dL 39 (SD 11.9), and working-lifetime integrated blood lead index was 719 μg·year/dL (SD 421.0) (Bleecker et al., 2003). After adjusting for the contribution of age, blood lead and weighted average blood lead were significantly associated with Wave I, while lifetime integrated blood index was significantly associated with Wave III and I-III interpeak interval. Four groups similar in age were created with increasing abnormalities based upon clinical cut-off scores for wave I latency and I-V interpeak interval. Blood lead, weighted average blood lead, and lifetime integrated blood index were all significantly higher in the group with prolonged Wave I and I-V interpeak interval compared to the group with normal BAEPs.

In summary, one detailed study found blood lead associated with prolonged VEPs with a threshold effect at 17-20 μg/dL. The four studies examining BAEPs and lead exposure consistently found prolonged interpeak latencies in the brainstem auditory pathway more strongly associated with cumulative or average blood lead exposure.

6.3.5.7 Postural Stability, Autonomic Testing, and Electroencephalogram (EEG) and Occupational Lead Exposure

Postural sway measures balance or steadiness on a force platform. It is a complex task that requires the integration of visual, vestibular, and peripheral sensory inputs, as well as motor output. No standard protocol is used across studies. Studies reviewed in this section are summarized in Annex Table AX6-3.17.

Postural sway was evaluated in 49 chemical workers exposed to lead stearate, with a mean blood lead of 18 μg/dL, a mean weighted average blood lead of 24 μg/dL and a mean cumulative blood lead of 391 μg·year/dL (Yokoyama et al., 1997). Twenty-three controls were also examined. After adjustment for covariates, a concentration-response relationship was observed

for blood lead and sway in the anterior-posterior direction and for weighted average blood lead with right to left sway. The authors concluded that change in the vestibulocerebellum was affected by blood lead while the anterior cerebellar lobe was affected by average lead exposure.

Chia et al. (1994) measured postural sway parameters in 60 lead storage battery workers (mean blood lead 36 μg/dL) and 60 controls (mean blood lead 6 μg/dL). Computerized postural sway measurements showed that lead workers had poorer postural stability that increased with eyes closed but no concentration-response association was observed with blood lead. A second publication examined cumulative blood lead over 10 years and found that lifetime integrated blood index for the 2 years prior to testing was associated with all postural sway parameters with eyes closed (Chia et al., 1996c).

Postural control measured in 63 lead battery workers (mean past blood lead 38 μg/dL) indicated significantly increased mean body oscillations with eyes closed and head tilted forward (Ratzon et al., 2000). Total lead exposure was significantly associated with increased body oscillations with head tilted forward after adjusting for education, coffee consumption, hours of sleep, and estimate of health. In order to maintain balance, lead-exposed workers required increased oscillations when visual and vestibular inputs were altered.

The effects of lead on the cardiac autonomic nervous system, expressed as the decrease of R-R interval variation on an electrocardiogram, was examined in 172 male lead-exposed workers (mean blood lead 36 μg/dL) (Teruya et al., 1991). A significant blood lead concentration-related decrease of R-R interval variation during deep breathing was present in 132 workers with stable blood lead over the past year. An approximate threshold effect was found at blood lead $\geq$20 μg/dL. Similar findings were reported by Niu et al. (2000) in 44 lead exposed workers who had a mean blood lead of 29 μg/dL.

One hundred twenty-eight workers in the ceramic painting industry (mean blood lead 13 μg/dL) were monitored for measures of sympathetic nerve function by variations in R-R interval on electrocardiography and changes in finger blood flow with postural changes using Doppler flowmetry (Ishida et al., 1996). No significant association was found between blood lead levels and the results of the neurophysiological tests, except for change in finger blood flow. Increased blood lead was associated with decreased changes in finger blood flow.

Sixty workers in a lead battery factory examined with quantitative electroencephalographs found alpha and beta frequencies were more abundant in workers with higher long term lead

exposure (Kovala et al., 1997). Biomarkers of long-term lead exposure included tibia bone lead (mean 26 μg/g), calcaneal bone lead (mean 88 μg/g), lifetime integrated blood index (mean 546 μg·year/dL), and weighted average blood lead (mean 32 μg/dL). The finding of slow alpha activity positively correlated with lead exposure may reflect increased episodes of "microdrowsiness" in workers with higher lead exposure. In the study by Niu et al. (2000), quantitative electroencephalographs in 44 lead-exposed workers (mean blood lead 29 μg/dL) indicated significantly increased beta activity and diminished amplitudes abnormalities in 81% of exposed workers.

In summary, postural sway is associated with lead exposure at blood lead levels <40 μg/dL, and is believed to be caused by the effect of lead on the cerebellum. A standard protocol was not employed across the studies. Parasympathetic and sympathetic integrity is compromised in lead-exposed workers beginning at blood lead >20 μg/dL. Quantitative electroencephalographs found increased beta activity associated with lead exposure.

6.3.5.8 Other Neurological Outcomes Associated with Lead in Adults

Studies reviewed in this section are summarized in Annex Table AX6-3.18. The 1986 Lead AQCD concluded that the evidence for an association of lead and ALS or motor neuron disease was inconsistent. The subsequent publications remain mixed but more studies are reporting an association. Using 109 cases of ALS and 256 controls matched for age, gender, and region of residence, Kamel et al. (2002) examined the relation of lead and ALS using blood lead and bone lead levels. Ranges of exposure were <1 to 14 μg/dL for blood lead, −4 to 107 μg/g for patella lead, and −7 to 61 μg/g for tibia lead. History of occupational lead exposure increased the risk of ALS (adjusted odds ratio of 1.9 [95% CI: 1.1, 3.3]). Elevations in both blood lead and patella and tibia bone lead were found in ALS cases, though the precision of these measurements was questioned. In summary, this study found lead exposure from historical questionnaire data and biological markers to be associated with ALS. The same data was used to determine the associations of ALS with polymorphism in ALAD and VDR and the influence of genotype in the previously discussed associations of ALS with lead (Kamel et al., 2003). The ALAD2 allele was associated with a 2-fold increased risk of ALS after adjustment for age, gender, region, education, and physical activity. Additionally adjusting for blood lead

strengthened the association of ALAD2 and ALS risk. This was not found for bone lead or occupational history of lead exposure. VDR was not associated with lead or ALS risk.

A study from the Mayo Clinic examined risk factors for sporadic ALS in 45 male ALS patient-patient control pairs (Armon et al., 1991). When lifetime exposure to lead exceeded 200 hours, the relative risk for ALS was 5.5 (95% CI: 1.44, 21.0). Overall, men with ALS had worked more at blue-collar jobs with significantly more time welding or soldering than controls ($p < 0.01$). The association between lead exposure and development of ALS was supported as these authors had the same findings in a previous pilot study of another patient population (Roelofs-Iverson et al., 1984).

Another study of risk factors for ALS in 103 patients found increased odds ratio for manual occupation (2.6 [95% CI: 1.1, 6.3]) and occupational exposure to lead (5.7 [95% CI: 1.6, 30]) (Chancellor et al., 1993). A Swedish study of 92 cases of motor neuron disease (includes ALS, progressive bulbar palsy, and progressive muscular atrophy) found a Mantel-Haenszel odds ratio for welding equal to 3.7 (95% CI: 1.1, 13.0) (Gunnarsson et al., 1992).

Guidetti et al. (1996) performed a retrospective incidence, prevalence, and mortality survey in northern Italy. The area studied had documented lead pollution for years. Based upon 79 cases, incidence and prevalence rates of ALS were comparable to the surrounding area. A subsequent publication by this group found that mean blood lead levels in cases of sporadic ALS and controls were not significantly different (mean blood lead of 13 µg/dL versus 11 µg/dL) (Vinceti et al., 1997). Blood lead was associated with disability due to ALS but no support was found for involvement of lead in the etiology of sporadic ALS.

Louis et al. (2003) examined the relationship between blood lead and essential tremor (ET) in 100 cases with ET (mean blood lead 3 µg/dL) and 143 controls (mean blood lead 2 µg/dL). Ten cases and 7 controls had bone lead levels measured that were significantly correlated with blood lead suggesting that higher blood lead may have occurred in the past. Logistic regression adjusting for age and current cigarette smoking found an association between blood lead and ET. An odds ratio of 1.19 (95% CI: 1.03, 1.37) was estimated. Blood lead was higher in the 39 ET cases with no family history. Both current and lifetime prevalence of occupational lead exposure was the same in ET cases and controls. In a second publication (Louis et al., 2005), 63 ET cases (mean blood lead 4 µg/dL) and 101 controls (mean blood lead 3 µg/dL) who were similar in age, education, gender, and ethnicity were examined for interaction

of blood lead and ALAD gene polymorphisms and increased odds of ET. Of the 63 ET cases, 18 (29%) had an ALAD2 allele compared to 17 (17%) of the 101 controls (odds ratio of 1.98 [95% CI: 0.93, 4.21]). When log blood lead was examined by presence of ALAD2 allele in ET, log blood lead was highest in ET cases with the ALAD2 allele, intermediate in ET cases without an ALAD2 allele, and lowest in controls (test for trend, $\beta = 0.10$; $p = 0.001$). When the ALAD2 allele was present, blood lead was significantly associated with odds of ET (80.29 [95% CI: 3.08, 2,096.36]). This increased odds of ET with an ALAD2 allele was 30 times greater than in individuals with only an ALAD1 alleles. In the highest log blood lead tertile, ALAD2 allele was present in 22% of ET cases and 5% of controls. It was proposed that increased blood lead along with the ALAD2 allele could affect the cerebellum and, thereby, increase the risk of tremor.

Graves et al. (1991) performed a meta-analysis on 11 case-control studies of Alzheimer's disease for occupational exposure to solvents and lead. Four studies had data for lead exposure with a pooled analysis of relative risks for occupational lead of 0.71 (95% CI: 0.36, 1.41). The exposure frequencies were 16 of 261 (6%) for the cases and 28 of 337 (8%) for the controls. These nonsignificant results were further confirmed by measuring lead concentration in the brain of cases with diffuse neurofibrillary tangles with calcification (DNTC), Alzheimer's disease, and non-demented controls. The lead concentration was significantly higher in DNTC compared to Alzheimer's disease and non-demented controls (Haraguchi et al., 2001).

In summary, more studies are reporting an association with past exposure to lead, usually in the occupational setting, and the motor neuron disease ALS. There appears to be a 2-fold increased risk for ALS when the ALAD2 allele is present. The odds of ET in individuals with the ALAD2 allele were 30 times greater compared to those with only ALAD1 alleles.

6.3.5.9 Occupational Exposure to Organolead and Inorganic Lead

Compared to inorganic lead, organolead exposure has a greater impact on the brain and, therefore, is discussed separately. Direct comparison of trimethyl lead (a metabolite of organolead), tetraethyl lead, and inorganic lead on the in vitro assembly of microtubules from the mammalian brain found no effects with inorganic lead but trimethyl lead produced dramatic impairment of neurotubular structures and functions (Roderer and Doenges, 1983). Another study examining organic and inorganic lead found differential effects on neurite growth in

neurons in culture, suggesting that the mechanism of action for organic and inorganic lead was not the same (Audesirk et al., 1989). Studies reviewed in this section are summarized in Annex Table AX6-3.19.

Two hundred and twenty-two current employees that manufactured tetraethyl lead had cumulative lead exposure associated with poorer performance in many cognitive domains but most often in manual dexterity and verbal memory/learning (Schwartz et al., 1993). Simple visual reaction time and blood lead had a curvilinear relation with an increase in simple visual reaction time occurring above a blood lead of 30 μg/dL (Balbus et al., 1997, 1998).

In former organolead workers (n = 543), peak tibial lead was a stronger predictor of poorer cognitive function than current tibial lead (Stewart et al., 1999). Examination of the peripheral nervous system in this population found no strong association between lead biomarkers and measures of sensory and motor function (Tassler et al., 2001). Five hundred and thirty-five of these former organolead workers were re-examined over a 4-year period (Schwartz et al., 2000d, 2001b). Peak tibia lead predicted decline in tests of verbal memory and learning, visual memory, executive ability, and manual dexterity. Effect size for an increase of 15.7 μg/g bone mineral of peak tibia lead was equivalent to 5 more years of age at baseline. This relationship of neurobehavioral tests with bone lead levels was influenced by the apolipoprotein E (*ApoE)* genotype (Stewart et al., 2002). The slope of the relation between tibia lead and neurobehavioral outcome was more negative in those individuals with at least one ε4 allele than individuals without this allele. It is suggested that the presence of one *Apo*-ε-4 allele increases the risk of persistent central nervous system effects of lead.

Overall when these neurobehavioral outcomes related to organolead exposure are compared to the literature reviewed with inorganic lead exposure, the absence of effects on the peripheral nerves and the global nature of central nervous system impairment suggests the impact on the brain is greater with organolead exposure.

6.3.6 Summary of the Epidemiologic Evidence for the Neurotoxic Effects of Lead in Adults

There is no consistent evidence that environmental lead exposure is associated with impaired cognitive performance in the elderly if competing risk factors are considered. In adults, the effect of lead on the nervous system may not be detected through neurobehavioral testing due

to cognitive reserve, the ability to compensate for brain impairment. Cognitive reserve is related to pre-morbid cognitive abilities, education, and occupational attainment, and is able to modify the clinical expression of central nervous system insult from lead exposure. Therefore, when chronic lead exposure is the same in two groups of individuals that differ by educational achievement levels, the concentration-response relationship will only be seen in the group with low educational achievement, as cognitive reserve allows the high educational achievement group to compensate for the central nervous system expression of the effects due to lead.

Chronic occupational lead exposure affects the sensory nerve fibers in the extremities with a possible threshold at a weighted average blood lead level of 28 μg/dL. Intensity of lead exposure appears to be more critical than duration of exposure for this outcome. Slowing in the brainstem auditory pathway in the caudal pons was consistently associated with chronic occupational lead exposure.

Past occupational exposure to lead increased the risk of developing ALS and motor neuron disease in 4 studies. This risk was increased 2-fold by the presence of the ALAD2 allele. Essential tremor in two well-done studies was associated with low blood lead levels (mean 3 μg/dL). The odds of developing ET with the ALAD2 allele increased 30-fold compared to those individuals with only an ALAD1 allele.

Numerous studies of occupational lead exposure also found chronic and current blood lead associated with visuomotor and memory impairment with a threshold effect at blood lead 18 μg/dL. As with ET, postural sway abnormalities associated with blood lead <40 μg/dL is believed to result from the effects of lead on different parts of the cerebellum.

6.4 RENAL EFFECTS OF LEAD

6.4.1 Summary of Key Findings on the Renal Effects of Lead from the 1986 Lead AQCD

Chronic lead nephropathy is a disease characterized by tubulointerstitial nephritis, which can ultimately result in small, fibrotic kidneys. It occurs in individuals who sustain chronic high-level lead exposure. In these individuals, lead exposure is the primary cause of renal failure. The pathophysiologic characteristics of lead nephropathy and the populations at increased risk for this diagnosis were the foci of the human research portion of Section 12.5, entitled “Effects

of Lead on the Kidney," in the 1986 Lead AQCD. The 1986 document clearly identified several high-risk groups for this diagnosis, including children in the Queensland, Australia lead poisoning epidemic, moonshine alcohol drinkers, and lead workers in poorly controlled settings. The section concluded that data in the latter group indicated an increased risk for lead nephropathy associated with blood lead levels ranging from 40 to >100 μg/dL, with adverse renal effects possibly occurring at levels as low as 30 μg/dL.

The 1986 Lead AQCD noted that research at that time was not sufficient to address some of the most critical questions relating to the impact of lead exposure on the kidney. The last paragraph of the renal section begins with "Among the questions remaining to be answered more definitively about the effects of lead on the kidneys is the lowest blood lead level at which renal effects occurs." The last sentence reads "Conversely, the most difficult question of all may well be to determine the contribution of low levels of lead exposure to renal disease of non-lead etiologies." Advances in the research conducted since that document was written allow a much more informed discussion of exactly those critical issues. As discussed below, recent research indicates that lead nephropathy is merely the tip of the iceberg in terms of the contribution that lead makes to renal dysfunction overall. Research increasingly indicates that lead, at much lower doses than those causing lead nephropathy, acts as a cofactor with other more established renal risks to increase the risk for renal dysfunction and the rate of subsequent decline. The populations at risk for renal dysfunction (diabetics and hypertensives) are increasing worldwide, particularly in countries where obesity is epidemic. Lead exposure is declining in many industrialized countries, although less so among high-risk minority populations. The extent of the public health impact of lead on the kidney depends on the balance of these two factors.

6.4.2 Renal Outcome Definitions

The renal literature can be confusing since several of the clinical renal measures are inversely related. Therefore, the pertinent outcomes are briefly reviewed below. The glomerular filtration rate (GFR) is considered to be the best measure of renal function. GFR is assessed by urinary clearance of exogenous (e.g., ^{125}I-iothalamate) or endogenous (e.g., blood urea nitrogen [BUN] and serum creatinine) compounds. Creatinine is used most commonly. Therefore, increases in BUN or serum creatinine or decreases in renal clearance of creatinine or other markers are all consistent with decreased renal function. Serum creatinine and its reciprocal

have been the most frequently used measures of renal function in the lead-kidney literature. However, creatinine is not an ideal GFR marker, because it is influenced by factors such as muscle mass, diet, gender, age, and tubular secretion. Measurement or calculation of creatinine clearance takes some of these variables into account. Measured creatinine clearance utilizes timed urine collections, traditionally over a 24-h period, making compliance difficult. Therefore, equations to estimate creatinine clearance have gained popularity. The Cockcroft-Gault equation (Cockcroft and Gault, 1976) has been used most commonly. Recently, several equations to estimate actual GFR were studied in the Modification of Diet in Renal Disease (MDRD) Study (Levey et al., 1999). The abbreviated MDRD equation (GFR in mL/min/1.73m^2 = 186 $\times$ creatinine$^{-1.154}$ $\times$ age$^{-0.203}$ $\times$ (0.742 if female) $\times$ (1.212 if African American); Stevens and Levey [2005a]) estimates GFR more accurately than the Cockcroft-Gault equation in patients with renal insufficiency (Levey et al., 2003). Despite their promise, however, the MDRD equations are relatively new and their use in the literature on the renal effects of lead exposure has been limited to date.

Cystatin C is another recent addition to the tools used to assess GFR (Stevens and Levey, 2005b). This is a 13,000 Dalton, non-glycosylated basic protein, which is generated by all nucleated cells and filtered, reabsorbed, and catabolized, but not secreted, in the kidney. Very little appears in the urine. The majority of studies done to date indicate that serum cystatin C is a better marker for GFR than serum creatinine (Stevens and Levey, 2005b).

Most of the renal outcome measures discussed above were developed for use in the clinical setting. Unfortunately, they are insensitive for early renal damage, as evidenced by the fact that serum creatinine remains normal after kidney donation. Therefore, in the last two decades, the utility of renal early biological effect (EBE) markers as indicators of preclinical renal damage has been of interest. These can be categorized as markers of function (i.e., low molecular weight proteins that should be reabsorbed in the proximal tubules such as β_2-microglobulin and retinol-binding protein [RBP]); biochemical alteration (i.e., urinary eicosanoids such as prostaglandin E_2, prostaglandin $F_{2\ alpha}$, 6-keto-prostaglandin $F_{1\ alpha}$, and thromboxane B_2); and cytotoxicity (e.g., N-acetyl-β-D-glucosaminidase [NAG]) (Cardenas et al., 1993). Elevated levels may indicate an increased risk for subsequent renal dysfunction. However, with the exception of microalbuminuria in diabetes and β_2-microglobulin in cadmium exposure, most are research tools only and their prognostic value remains controversial.

European and Asian nephrotoxicant researchers have utilized them more frequently than have renal researchers in the United States. Prospective studies of most of these markers in nephrotoxicant-exposed populations are quite limited to date.

6.4.3 Lead Exposure Measure Definitions

Although these definitions are reviewed in detail elsewhere in this Lead AQCD, a brief discussion is included here due to the number of key studies in this section that measured bone or chelatable lead dose. Inorganic lead is a cumulative toxicant that is stored in bone. Blood lead is a relatively short-term measure (half-life of 30 days [Hu et al., 1998]) that reflects exposure from current exogenous sources and the release of lead from internal lead stores. Bone is a source of lead as well as a repository (Hu et al., 1998). As such, bone lead measures provide information on the potential for ongoing internal exposure as well as cumulative exposure. Lead in trabecular bone (commonly measured in the patella or calcaneus) is more bioavailable than lead in cortical bone (measured in the mid-tibia) and has a shorter half-life (Gerhardsson, et al., 1993; Hu et al., 1998). An additional lead measure, chelatable lead, is thought to represent a bioavailable pool of lead from blood, soft tissue, and bone. Either calcium disodium ethylenediaminetetraacetic acid (EDTA) or dimercaptosuccinic acid (DMSA; succimer) may be used for this purpose although DMSA is newer and, thus, has been used less frequently to date.

6.4.4 Lead Nephrotoxicity in Adults

6.4.4.1 General Population Studies

Over the past two decades, several studies have examined the effect of lead exposure on renal function in environmentally exposed general populations. This is a new category of lead-renal research with no high quality examples (by current standards) having been available for review in the 1986 Lead AQCD. The studies discussed below provide critical evidence that the adverse effects of lead on the kidney occur at much lower doses than previously appreciated. General population studies of the renal effects of lead are further summarized in Annex Table AX6-4.1.

6.4.4.1.1 Cadmibel Study

In the first large environmental study that adjusted for multiple renal risk factors, Staessen et al. (1992) evaluated 965 men and 1,016 women in the Belgian Cadmibel study. Lead dose was indexed by blood lead and zinc protoporphyrin. Renal outcome measures included serum creatinine and β_2-microglobulin and 24-h measured and calculated (Cockcroft and Gault, 1976) creatinine clearances. Mean blood lead was 11.4 μg/dL (range 2.3-72.5) and 7.5 μg/dL (range 1.7-60.3) in men and women, respectively. After adjustment, log transformed blood lead and zinc protoporphyrin, in separate models, were negatively associated with measured creatinine clearance (effect estimates are presented in Table 6-4.1). A 10-fold increase in blood lead was associated with a decrease in creatinine clearance of 10 and 13 mL/min in men and women, respectively. Both lead measures were also negatively associated with estimated creatinine clearance. This landmark study raised concern that the lead dose threshold for adverse renal effects in the general population was much lower than previously appreciated based on occupational data.

6.4.4.1.2 Normative Aging Study

Four studies assessing the renal impact of lead exposure in the Normative Aging Study have been published to date. Participants in this study were originally recruited in the 1960s in the Greater Boston area. Inclusion criteria included male gender, age between 21 and 80 years, and absence of chronic medical conditions. Payton et al. (1994) analyzed data from a periodic follow-up evaluation performed between 1988 and 1991 in 744 participants. Lead dose was assessed with blood lead; renal outcome measures included serum creatinine and 24-h measured and calculated (Cockcroft and Gault, 1976) creatinine clearances. Mean blood lead concentration and measured creatinine clearance were 8.1 μg/dL (SD 3.9) and 88.2 mL/min (SD 22.0), respectively. After adjustment, ln blood lead was negatively associated with ln measured creatinine clearance (effect estimates are presented in Table 6-4.1). Borderline statistically significant associations ($p < 0.1$) between blood lead and serum creatinine and estimated creatinine clearance were also observed.

Kim et al. (1996) studied 459 men whose blood lead levels from past periodic examinations, conducted every 3-5 years during 1979-1994, were measured from stored samples. Participants were randomly selected to be representative of the entire Normative Aging Study

Table 6-4.1. Summary of Key Studies on the Renal Effects of Environmental Lead Exposure

Reference Study location Study population Sample size	Mean exposure and outcome measures	Analysis methods Covariates adjusted for in analysis	Major significant findings
Muntner et al. (2003) NHANES III, 1988-1994 n = 15,211 4,813 hypertensives	Blood lead 4.21 µg/dL (hypertensives) 3.3 µg/dL (normotensives) Renal outcomes = elevated serum creatinine, chronic kidney disease (GFR <60 mL/min/1.73 m^2)	Multiple logistic regression Age, race, gender, diabetes, systolic blood pressure, smoking status, history of cardiovascular disease, body mass index, alcohol consumption, household income, marital status, and health insurance	Higher odds ratios of both increased serum creatinine and chronic kidney disease by quartile of blood lead in hypertensives but not normotensives Odds ratios for elevated serum creatinine in hypertensives Blood lead (range, µg/dL) / % / Odds ratio (95% CI) Quartile 1 (0.7–2.4) / 7.2 / 1.00 Quartile 2 (2.5–3.8) / 12.1 / 1.47 (1.03, 2.10) Quartile 3 (3.9–5.9) / 12.4 / 1.80 (1.34, 2.42) Quartile 4 (6.0–56.0) / 16.3 / 2.41 (1.46, 3.97) $p < 0.001$ for chi-squared test for trend Twofold higher blood lead associated with odds ratio of 1.43 (95% CI: 1.20, 1.71)
Payton et al. (1994) Boston, MA Normative Aging Study, 1988-1941 n = 744	Blood lead 8.1 µg/dL Measured creatinine clearance 88.2 mL/min	Multiple linear regression Age, body mass index, analgesic and diuretic use, alcohol consumption, smoking status, systolic/ diastolic blood pressure	Log blood lead negatively associated with log measured creatinine clearance −0.04 (95% CI: −0.079, −0.001) 10 µg/dL higher blood lead associated with a 10.4 mL/min lower creatinine clearance
Kim et al. (1996) Boston, MA Normative Aging Study, 1979-1994 n = 459	Blood lead at baseline 9.9 µg/dL Serum creatinine at baseline 1.2 mg/dL	Cross-sectional and longitudinal analyses Random-effects modeling Baseline age, time since initial visit and between visits, body mass index, smoking status, alcohol ingestion, education level, hypertension (defined as blood pressure ≥160 or 95 mm Hg or antihypertensive medication use), and baseline serum creatinine	In cross-sectional analyses of associations between log transformed blood lead and concurrent serum creatinine, the largest β was in the 141 participants whose peak blood lead ≤10 µg/dL: 0.06 (95% CI: 0.023, 0.097) Positive association between log transformed blood lead and change in serum creatinine over subsequent follow-up period in participants whose peak blood lead was ≤25 µg/dL 0.027 (95% CI: 0.0, 0.054)

Table 6-4.1 (cont'd). Summary of Key Studies on the Renal Effects of Environmental Lead Exposure

Reference Study location Study population Sample size	Mean exposure and outcome measures	Analysis methods Covariates adjusted for in analysis	Major significant findings
Wu et al. (2003) Boston, MA Normative Aging Study, 1991-1995 n = 709	Blood lead 6.2 µg/dL Patella lead 32.1 µg/g bone Calculated creatinine clearance 71.3 mL/min	Multiple linear regression Age, body mass index, hypertension, smoking status, alcohol ingestion, analgesic medication use	Significant association between patella lead and creatinine clearance $\beta = -0.069$ (SE not provided)
Tsaih et al. (2004) Boston, MA Normative Aging Study 1991-~2001 n = 448	Blood lead at baseline 6.5 µg/dL Tibia lead at baseline 21.5 µg/g bone mineral Serum Creatinine at Baseline 1.3 mg/dL	Longitudinal analysis, mean of 6 years between evaluations Age, body mass index, diabetes, hypertension, smoking status, alcohol consumption, analgesic use, baseline serum creatinine and its square	Lead dose not associated with change in creatinine in all Significant interaction of blood and tibia lead with diabetes in predicting annual change in serum creatinine For natural ln baseline blood lead $\beta = 0.076$ (95% CI: 0.031, 0.121) compared to $\beta = 0.006$ (95% CI: −0.004, 0.016) for non-diabetics For natural ln baseline tibia lead $\beta = 0.082$ (95% CI: 0.029, 0.135) compared to $\beta = 0.005$ (95% CI: −0.005, 0.015) for non-diabetics
Staessen et al. (1992) Belgium Cadmibel Study n = 1,981; 965 males	Blood lead 11.4 µg/dL (males) 7.5 µg/dL (females) Measured creatinine clearance 99 mL/min (males) 80 mL/min (females)	Multiple linear regression Age, age squared, body mass index, log transformed gamma-glutamyl transpeptidase, and diuretic use	Log transformed blood lead negatively associated with measured creatinine clearance −9.5 (95% CI: −18.1, −0.9) males −12.6 (95% CI: −20.3, −5.0) females Tenfold increase in blood lead associated with a decrease in creatinine clearance of 10 and 13 mL/min in men and women, respectively

population in terms of age and follow-up. Renal status was assessed with serum creatinine. Data from 4-5 evaluations were available for the majority of participants. Relations were evaluated cross-sectionally (associations between blood lead and concurrent serum creatinine) as well as longitudinally (associations between blood lead and change in serum creatinine over the subsequent follow-up period). Mean age, blood lead level, and serum creatinine, at baseline, were 56.9 years (SD 8.3), 9.9 μg/dL (SD 6.1), and 1.2 mg/dL (SD 0.2), respectively. With random-effects modeling, a significant positive association between ln-transformed blood lead and concurrent serum creatinine was observed. This association was stronger when models were confined to participants with lower peak blood lead levels, i.e., the β coefficient was largest in the 141 participants whose highest blood lead level was ≤10 μg/dL. In longitudinal analysis, ln-transformed blood lead was associated ($p = 0.05$) with change in serum creatinine over the subsequent follow-up period in the 428 participants whose highest blood lead level was ≤25 μg/dL. Similar to the cross-sectional analysis, the β coefficient in the participants whose highest blood lead level was ≤10 μg/dL was larger; however, in the longitudinal analysis, the standard error also increased such that the p-value was not significant.

Cortical and trabecular bone lead measurements were obtained in evaluations performed between 1991 and 1995 in 709 participants in the Normative Aging Study (Wu et al., 2003). Lead dose was assessed with blood, tibia, and patella lead concentrations. Renal outcome measures included serum creatinine and estimated creatinine clearance. Mean blood, tibia and patella lead levels were 6.2 μg/dL (SD 4.1), 22.0 μg/g bone mineral (SD 13.4), and 32.1 μg/g bone mineral (SD 19.5), respectively. After adjustment, analyses in the 670 participants from whom these data were available, revealed a significant inverse association between patella lead and creatinine clearance. A borderline significant ($p = 0.08$) inverse association between tibia lead and creatinine clearance was also observed. None of the lead measures were significantly associated with serum creatinine.

Tsaih et al. (2004) reported associations between baseline lead dose and change in serum creatinine in 448 men. Lead dose was assessed with blood, tibia, and patella lead. Serum creatinine was measured at baseline and at follow-up, an average of 6 years later. Six percent and 26% of subjects had diabetes and hypertension, at baseline, respectively. Mean blood lead levels and serum creatinine decreased significantly over the follow-up period in the group. Lead dose was not associated with change in creatinine in all participants. However, the authors found

a significant interaction between lead dose (blood and tibia lead) and diabetes on change in serum creatinine. Interaction was also observed between tibia lead and hypertension, although it is possible that many of the 26 diabetics were also included in the hypertensive group and were influential there as well.

6.4.4.1.3 NHANES III

Muntner et al. (2003) analyzed associations between blood lead and renal outcomes in 15,211 adult subjects enrolled in the NHANES III study, conducted from 1988 through 1994. Dichotomous renal outcome measures analyzed included elevated serum creatinine and chronic kidney disease (GFR < 60mL/min/1.73 m^2). Due to interaction between blood lead and hypertension, the population was stratified. Mean blood lead was 4.21 μg/dL in the 4,813 hypertensives and 3.30 μg/dL in normotensives. The prevalence of elevated serum creatinine in hypertensives and nonhypertensives was 11.5% and 1.8%, respectively; prevalence of chronic kidney disease was similar. The odds ratios for both renal outcomes increased by quartile of blood lead among the hypertensive subjects but not among those without hypertension. Among those with hypertension, after adjustment for age, race and gender, the odds ratios for elevated creatinine in quartiles 2, 3, and 4 compared to the lowest quartile of blood lead, were 1.56 (95% CI: 1.04, 2.35), 1.68 (95% CI: 1.24, 2.26), and 2.07 (95% CI: 1.26, 3.40), respectively. As shown in Table 6-4.1, the odds ratios were the same following additional adjustment. The authors noted that the "associations were strong, dose-dependent and consistent before and after comprehensive adjustment." They also noted that in nonhypertensives, higher blood lead was associated with a higher prevalence of chronic kidney disease in diabetics. This study is notable for the sample size, for the reported associations being observed at the lowest mean blood lead level in any environmental study to date, for the comprehensive adjustment for other renal risk factors, and for the study population being representative of the U.S. population.

6.4.4.1.4 Summary of Lead Nephrotoxicity in the General Population

Studies of environmentally exposed general populations constitute one of the two most important types of research on the adverse renal effects of lead during the past two decades. Study designs are generally strong; some have the added strength of analyzing longitudinal data. Populations are large, assessment of lead dose is comprehensive, including the use of bone lead

as a measure of cumulative lead body burden in some studies, and statistical approaches are advanced, utilizing a range of exposure and outcome measures, while adjusting for numerous renal risk factors. Given these strengths, the fact that these studies have reached consistent conclusions provides strong evidence indicating that lead is a contributor to renal dysfunction in susceptible populations at much lower levels that those identified in the 1986 Lead AQCD. Chronic kidney disease has been observed at the lowest lead dose levels studied (category II from 2.5 to 3.8 μg/dL in Muntner et al. [2003]). An association between cumulative lead dose (mean tibia lead of 21.5 μg/g bone mineral) and longitudinal decline in renal function has been observed as well, although data on any threshold for this effect were not reported (Tsiah et al., 2004). Susceptible populations include those with other risk factors for renal disease, including hypertension and diabetes. Populations who are also at increased risk for obesity, diabetes, and hypertension represent groups potentially most impacted by lead exposure.

6.4.4.2 Occupational Studies

The vast majority of studies in the lead-renal literature were conducted in the occupational setting. This was especially true prior to the 1986 Lead AQCD but is still currently the case. Occupational studies of the renal effects of lead are presented in Annex Table AX6-4.2. Recent studies in the general population, discussed above, and in the patient population, discussed in the next section, provide consistent evidence supporting a role for lead in renal dysfunction at lower lead concentrations of interest. However, one phenomenon that has been observed more frequently in occupational rather than environmental studies of lead exposure and kidney function deserves specific comment.

Several studies have reported statistically significant negative associations between higher lead dose and worse renal function, specifically positive associations between higher lead dose and lower BUN, serum creatinine and/or higher creatinine clearance. Roels et al. (1994) observed higher mean creatinine clearance in 76 lead workers compared to 68 controls from the same smelter who were not occupationally exposed to lead (mean of 121.3 versus 115.5 mL/min/1.73 m^2 in workers and controls, respectively [$p < 0.05$]). More importantly, in the combined group, tibia lead was positively correlated with measured creatinine clearance. However, no other significant associations between lead dose and the renal outcomes (which also included serum creatinine, urea nitrogen, and β_2-microglobulin, along with urinary NAG, RBP

and β_2-microglobulin as well as other early biological effect markers) were observed. Lead workers had evidence of high past exposure and controls also had high blood lead levels by current standards (mean blood and tibia lead levels were 43.0 and 14.1 µg/dL and 66 and 21 µg/g bone mineral, in workers and controls, respectively).

Weaver et al. (2003a) performed a cross-sectional analysis of first evaluation data from a longitudinal study of 803 lead workers in South Korea, including 94 former lead workers. Lead exposure was assessed with job duration; blood, tibia, and DMSA-chelatable lead; and three hematologic measures as surrogates for lead dose. Clinical renal function was assessed with blood urea nitrogen (BUN), serum creatinine, measured creatinine clearance, and calculated creatinine clearance (Cockcroft and Gault, 1976). Urinary NAG and RBP were also measured. Mean job duration, and blood, tibia, and DMSA-chelatable lead levels were 8.2 years (SD 6.5), 32.0 µg/dL (SD 15.0), 37.2 µg/g bone mineral (SD 40.4), and 767.8 µg/g creatinine (SD 862.1), respectively. Higher lead measures were associated with worse renal function in nine of the 42 associations, however, an additional five were in the opposite direction (higher lead measures associated with lower serum creatinine and higher creatinine clearances). These opposite direction (inverse) associations were observed only for the clinical outcomes whereas the associations between higher lead dose and worse renal function were predominantly with the EBE markers. In three of 16 models (analyses) assessing effect modification by age on associations between lead job duration and the three lead dose biomarkers with the four clinical renal outcomes, positive associations between higher lead measures and worse renal function in participants in the oldest age tertile were significantly different from associations in those in the youngest age tertile, which were in the opposite (inverse) direction; this pattern was observed at borderline significance ($p < 0.1$) in three other models. However, this pattern was not observed in the EBE marker models. Similar inverse associations were observed in this population in the third evaluation, performed a mean of 2.2 years after collection of the data discussed above, but only with DMSA-chelatable lead and not patella, blood, or tibia lead (Weaver et al., 2005b). Hsiao et al. (2001) also reported positive associations between higher blood lead and lower concurrent serum creatinine in an analysis of 8 years of annual medical surveillance data in 30 lead battery workers (this study is of note since it is one of the few longitudinal, occupational studies to date; additional findings are described in Annex Table AX6-4.2).

These inverse associations, evidenced by higher lead measures and lower BUN and serum creatinine and/or higher creatinine clearance, may represent lead-induced hyperfiltration, a phenomenon initially observed in patients with diabetes but also implicated in other settings, including hypertension and obesity (Nenov et al., 2000). In this process, initial supranormal renal function is paradoxically associated with increased risk for subsequent renal dysfunction. Hu (1991) has also reported increased mean creatinine clearance in 22 adults who were lead poisoned as children compared to matched controls. Longitudinal data for lead-exposed rodents (discussed in Section 5.7) are critical in relating this process to lead. However, in that work, despite similar initial hyperfiltration, subsequent renal dysfunction was much more severe in the high-dose lead-exposed rodents compared to the low-dose animals. This suggests that hyperfiltration may be one, but not the only, mechanism for the adverse renal effects of lead. Whether hyperfiltration contributes to pathology in humans is unclear; longitudinal studies are needed.

Regardless, the issue for this document is that significant findings could be obscured if opposite direction associations are present in different segments of the study population and interaction models to address this are not performed. This is a valid concern, since the factors involved in these inverse associations in lead exposed populations are not well defined at present. Work by Weaver and colleagues have used age as the effect modifier; however, other factors, such as lead job duration, may be more important modifiers.

Figure 6-4.1 provides an example of the different associations observed depending on whether effect modification is examined. In the work of Weaver et al. (2003a), in several models, no associations were observed when the entire population was studied; however, when interaction models using age as the effect modifier were evaluated, significant associations in opposite directions were observed.

6.4.4.3 Patient Population Studies

Studies in various patient populations have also contributed to the body of knowledge concerning adverse renal impacts of lead exposure. Such studies of renal effects of lead in patient populations are presented in Annex Table AX6-4.3. Populations studied include those with chronic renal insufficiency (CRI), end-stage renal disease (ESRD), gout, and hypertension. Patients were selected for study due to the fact that these diseases are thought to be increased by

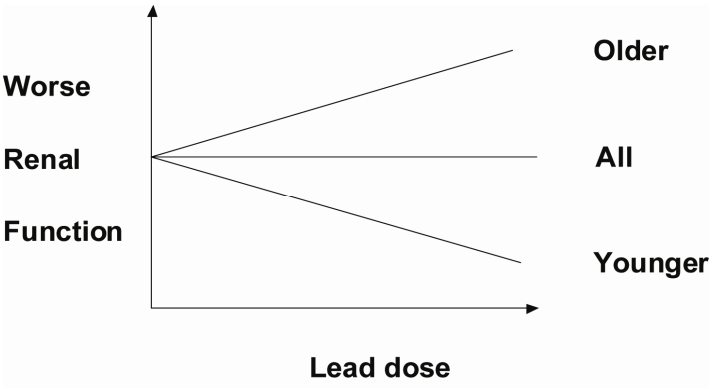

Figure 6-4.1. Effect on renal function evaluation using age as the effect modifier.

Source: Weaver et al. (2003a).

high-level lead exposure, particularly when two or more coexist in the same patient. Early work in this area was discussed in the 1986 Lead AQCD, but some research is discussed again below in order to allow conclusions to be drawn on the data to date. Early research focused on patients with potential lead nephropathy; and lead body burdens of interest, assessed with EDTA chelation, were above 600 to 650 µg/72 h. More recent work has involved patients with CRI but not ESRD and lead body burdens below that range. Of note, the terminology for degree of renal dysfunction was more variable in the older literature. Therefore, for clarity in the discussion below, patients with impaired renal function who are not on dialysis are noted to have CRI, rather than using terms such as "failure" that were used in some of the original reports.

Two issues have been a recurring concern in this work, particularly in work with patients on dialysis. The first concern is whether lead body burden is higher in all patients with renal insufficiency or failure due to decreased lead excretion (reverse causality). The second concern is whether EDTA-chelatable lead levels when measured over a 72-h period in patients with CRI can be equated to those in participants with normal renal function measured over 24 h. It is possible that, due to decreased excretion of EDTA in renal insufficiency, more lead per dose is ultimately chelated. These concerns have been addressed in various ways as noted in the research discussed below. Lastly, this work also illustrates the limitations of blood lead levels which often do not reflect the differences in lead body burden noted between populations.

6.4.4.3.1 Lead Body Burden in Chronic Renal Disease

Batuman et al. (1983) studied 27 hypertensives with CRI (defined as serum creatinine >1.5 mg/dL) and 21 without associated renal impairment. Blood lead levels were similar in the two groups. However, mean EDTA-chelatable lead levels were significantly higher in hypertensives with CRI than those without (860 and 340 μg/72 h, respectively). Further, chelatable lead levels in patients with CRI from causes not thought to be related to lead nephropathy and who had no history of lead exposure were similar to patients with hypertension but no CRI. This provides some evidence against reverse causality.

Sanchez-Fructuoso et al. (1996) performed a similar study in a much larger number of patients in Spain, none of whom had a known history of lead exposure. These authors reported that EDTA-chelatable lead levels >600 μg/72 h were present in none of 30 controls, 16 (15.4%) of 104 patients with essential (primary) hypertension and normal renal function, 74 (56.1%) of 132 patients with CRI of unknown etiology along with hypertension (64 of the 132 also had gout), but none of the 30 patients with CRI of known (non-lead related) etiology. Mean blood and EDTA-chelatable lead levels in the patients with CRI of known cause were not statistically different from controls with normal renal function. These researchers also reported significant correlations between bone lead levels (assessed by biopsy) and EDTA-chelatable lead level in 12 patients whose chelatable lead levels were >600 μg/72 h, which provides support for the validity of chelatable lead levels in CRI.

In contrast, Osterloh et al. (1989) reported no significant difference in EDTA-chelatable lead levels between 40 male subjects with hypertensive nephropathy (hypertension preceded renal insufficiency; serum creatinine 1.8-4 mg/dL) and 24 controls with renal dysfunction from other causes. Lead dose and serum creatinine were not correlated. Chelatable lead levels in this population were much lower than those reported by Wedeen et al. (1983) and Sanchez-Fructuoso et al. (1996). The authors noted that only 17% of their study participants had a history of possible lead exposure based on questionnaire. In contrast, Batuman et al. (1983) found that 89% of patients with hypertension and CRI had a possible history of lead exposure. The inconsistent results in these studies may reflect differences in the patients studied. Batuman et al. (1983) studied Veterans Administration patients, Sanchez-Fructuoso et al. (1996) studied patients from a low-medium income area in Madrid, Spain, and Osterloh et al. (1989) recruited patients from the database of a large health maintenance organization in California.

Van de Vyver et al. (1988) reported lead data from bone biopsies in 153 dialysis patients, 11 cadavers without known excessive lead exposure, 13 patients with renal insufficiency, gout, and/or hypertension and 22 lead workers. Bone lead levels in 5% of the dialysis population were in the range observed in lead workers, suggesting lead as a primary cause of their renal failure. Levels in the 10 patients with analgesic nephropathy were the lowest (all <7 μg/g). However, Winterberg et al. (1991) subsequently noted that the bone lead levels in patients with analgesic nephropathy and cadaver controls in Van de Vyver et al. (1988) were much higher than in control groups of other researchers. They reiterated the concern that lead did accumulate due to decreased renal excretion. In a longitudinal study, Price et al. (1992) reported similar half-lives of lead in bone in eight renal patients compared with age-matched controls who had XRF finger bone lead conducted twice 5 years apart. The small number and inclusion of outliers without formal statistical analysis limits conclusions that can be drawn from these data. The longitudinal studies of Lin and colleagues, discussed below, provide more definitive data in this regard.

6.4.4.3.2 Impact of Lead Body Burden on Decline in Renal Function in Patients with CRI

Lin and colleagues have addressed the issue of low-level lead as a cofactor with other renal risk factors in susceptible populations, including those with CRI and/or gout. They have approached this work in two ways: prospective follow-up of populations with CRI to determine if renal function decline is greater in those with higher lead body burdens and through randomized trials to determine if chelation therapy changes the rate of renal function decline. Importantly, their work is in an EDTA-chelatable lead range well below that considered abnormal as described in Section 6.4.4.3.1.

In their most recent publication, Yu et al. (2004) followed 121 patients over a 4-year period. Eligibility required well-controlled CRI. Importantly, serum creatinine between 1.5 and 3.9 mg/dL and EDTA-chelatable lead <600 μg/72 h were required at baseline. Patients with potentially unstable renal disease were excluded (i.e., due to systemic diseases such as diabetes). Sixty-three patients had "high-normal" EDTA-chelatable lead levels (≥80 but <600 μg/72 h); 58 had "low-normal" EDTA-chelatable lead levels (<80 μg lead/72 h). The groups were similar in most other baseline risk factors. Borderline statistically significant ($p < 0.1$) differences included mean older age in the high chelatable lead group and certain renal diagnoses. Fifteen patients in the "high-normal" chelatable lead group reached the primary endpoint (doubling of

serum creatinine over the 4-year study period or need for hemodialysis) compared to only two in the "low-normal" group ($p = 0.001$).

In a Cox multivariate regression analysis, chelatable lead was significantly associated with overall risk for the primary endpoint (hazard ratio for each 1 µg chelatable lead was 1.01 [95% CI: 1.00, 1.01; $p = 0.002$]). In this model, the only other variable reaching at least borderline significance ($p < 0.1$) was baseline serum creatinine. The associations between baseline chelatable lead or blood lead level and change in GFR (estimated by an MDRD equation [Levey et al., 1999]) were modeled separately using GEE. Based on these models, a 10 µg higher chelatable lead level or a 1 µg/dL higher blood lead level reduced the GFR by 1.3 and 4.0 mL/min, respectively, during the 4-year study period. Similar to the primary outcome analysis, of the many traditional renal risk factors adjusted for in these models, only diagnosis of chronic interstitial nephritis was significantly associated, in this case with an increase in GFR. Of note, chronic interstitial nephritis was also a more frequent diagnosis in the group with the low-normal chelatable lead levels ($p = 0.09$).

6.4.4.3.3 Therapeutic EDTA Chelation in Patients

Chelation in lead exposure is controversial due to the potential for it to be used in lieu of exposure reduction. Chelation in lead nephropathy, in particular, is controversial, because cases of acute tubular necrosis were reported following early clinical use of EDTA that involved large doses in the treatment of hypercalcemia and lead poisoning. Adverse renal effects have not been observed in subsequent work using much lower doses (Sanchez-Fructuoso et al., 1996; Wedeen et al., 1983).

Work prior to the 1986 Lead AQCD suggested that chelation might be beneficial in lead nephropathy (Morgan, 1975; Wedeen et al., 1979). This issue has been addressed more recently by Lin and colleagues in patients with much lower lead doses. Lin et al. (1999) studied 43 patients with serum creatinine and EDTA-chelatable lead levels between 1.5-4 mg/dL and 150 and 600 µg/72 h, respectively. Patients were followed for 12 months to determine their baseline rate of renal function decline. A group of 32 was then randomized; and 16 underwent a 2-month treatment period consisting of weekly chelation with 1 g EDTA; whereas the other 16 continued their regular care. Traditional renal risk factors, such as blood pressure control, were similar in the two groups. Prior to therapeutic chelation, the rate of progression of renal

insufficiency was not statistically different. However, actual improvement in renal function was noted in the treated group during chelation and subsequent renal function decline was slower in this group. The mean difference in the change in the reciprocal of serum creatinine post therapy was 0.000042 L/μmol per month (95% CI: 0.00001, 0.00007).

In subsequent work, Lin et al. (2003) published results of a randomized chelation trial in a larger group. This work included a 2-year prospective study of renal function decline prior to chelation in 202 patients with CRI and EDTA-chelatable lead <600 μg/72 h. Results of the Cox proportional-hazards model were similar to those reported in Yu et al. (2004). Associations between baseline EDTA-chelatable lead level and change in GFR were modeled using GEE. After adjustment, an increase of 10 μg in EDTA-chelatable lead was associated with a GFR decrease of 0.03 mL/min/1.73 m^2 of body-surface area during the observation period ($p < 0.001$). Of note, this effect, although statistically significant, is 40-fold lower than that reported in Yu et al. (2004) over a follow-up period that is only 2-fold shorter. At 24 months, 64 patients whose EDTA-chelatable lead levels were 80-600 μg/72 h were randomized; half to a 3-month treatment period consisting of weekly chelation with 1 g EDTA until their excreted lead levels fell below 60 μg/72 h and half to placebo infusion over 5 weeks. Renal risk factors were similar in the two groups. Mean blood lead levels were 6.1 μg/dL and 5.9 μg/dL in treated and control groups, respectively. In the subsequent 24 months, chelation in 19 (59%) participants was repeated due to increases in serum creatinine in association with rebound increases in EDTA-chelatable lead levels. Each received one additional chelation series (mean = 4.1 g EDTA) a mean of 13.7 months after the first chelation period. At the end of the study period, mean estimated GFR increased by 2.1 mL/min/1.73 m^2 of body-surface area in the chelated group compared to a decline of 6.0 mL/min/1.73 m^2 of body-surface area in the controls ($p < 0.01$) (see Figure 6-4.2). The 95% CI for the difference between the chelated and control groups was −11.0 to −5.1 mL/min/1.73 m^2 of body-surface area.

Lin and colleagues have also reported chelation results in patients with gout. Historically, gout was known to be a risk from high-level lead exposure such as in the Queensland, Australia epidemic and in moonshine alcohol drinkers (U.S. Environmental Protection Agency, 1986a). Higher EDTA-chelatable lead levels in patients with both gout and CRI compared to those with CRI or gout alone have also been reported in several studies at lower levels of exposure

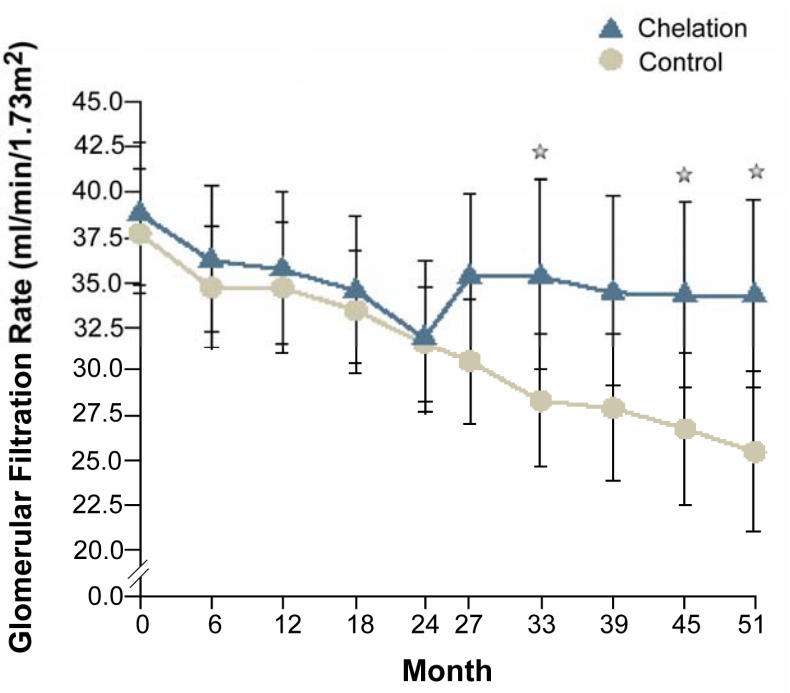

Figure 6-4.2. Estimated mean (±2 SE) glomerular filtration rate according to time in the chelation group (n = 31) and the control group (n = 30) during the observation and intervention periods. The patients in the chelation group received chelation therapy from month 24 to month 51. The asterisks indicate p < 0.05 by Student's t-test.

Source: Lin et al. (2003).

(Batuman et al.,1981; Sanchez-Fructuoso et al., 1996; Lin et al., 2001). Lin and colleagues have pursued this finding in chelation studies of patients with CRI with and without gout (2001) and in otherwise healthy patients with and without gout (2002). Both studies reported significant associations between EDTA-chelatable lead and uric acid measures before chelation and improvement in urate clearance and other uric acid measures after chelation (see Annex Table AX6-4.3 for data). In contrast, Miranda-Carus et al. (1997) did not find correlations between EDTA-chelatable lead body burdens above or below 600 µg/120 h and uric acid measures. Uric acid parameters were unchanged following chelation, although only six participants were studied and all had EDTA-chelatable lead above 600 µg/120 h. Therefore, lack of power as well as different lead body burdens may be explanatory factors for differences between this work and the Lin studies.

The key studies in patients followed prospectively with and without chelation constitute the other major advance in research on the adverse renal effects of lead over the past two decades. This work suggests that lead is nephrotoxic in susceptible populations at lower levels than currently appreciated. Blood lead levels (mean = 4.2 µg/dL [range 1.0–13.4]) and body lead burdens (mean = 99.1 µg/72 h [range 2.5–530]) were associated with decline in GFR over a 4-year follow-up period (Yu et al., 2004). Chelation was beneficial in a body lead burden range from 80 or 150 to 600 µg/72 h, depending on study (Lin et al., 2003; 1999). The published data do not allow a determination of whether a threshold exists. It is also possible that chelation has a direct beneficial effect on kidney function, regardless of lead exposure, since DMSA has been reported to prevent renal damage in a non-lead exposed rat model of nephrosclerosis (Gonick et al., 1996). If so, the benefits of chelation do not appear to occur via reversal of structural damage (Khalil-Manesh et al., 1992b); improved hemodynamics from reduction of reactive oxidant species may be a mechanism (Gonick et al., 1996).

Strengths of the work of Lin and colleagues include prospective study design, lead dose assessment including bioavailable body burden, statistical analysis that includes GEE for longitudinal data, and adjustment for more renal risk factors than any of the other key studies discussed in Section 6.4. Limitations include that fact that, to date, this type of research has been conducted in relatively small number of participants and in only one center. As noted above, the two reported lead body burden β coefficients in GEE models of decline in renal function vary widely. Therefore, small study sizes and differences in renal diagnoses between groups may be overly influential in the results. However, if confirmed in large populations, the potential public health benefit could be substantial. Therapeutic options would be available for high-risk patients, who, despite dramatic reductions in lead exposure in developed countries, are still adversely affected by lead. Lin et al. (2003) noted that, based on their data, chelation could delay the need for hemodialysis by 3 years. Therefore, this unique line of research is deserving of further study. Prospective studies consistent with the results of Yu et al. (2004) will likely be needed to justify randomized, controlled chelation trials.

6.4.4.4 Mortality Studies

As summarized in Steenland et al. (1992), mortality studies have consistently shown excess mortality from chronic kidney disease in lead workers. This increased risk has been most

apparent in workers exposed in earlier time periods, becoming nonsignificant in later calendar time periods in a number of studies. Steenland et al. (1992) reported similar results in a study of 1990 former lead smelter workers. This cohort was made up of predominantly white men who had worked in a lead-exposed department for at least 1 year between 1940 and 1965. Mean (SD) blood lead, measured in 1976 in 173 members of this cohort, was 56.3 µg/dL (12.9). There were 8 deaths from chronic kidney disease. Compared to the U.S. white male population, the standardized mortality ratio was 1.26 (95% CI: 0.54, 2.49). The standardized mortality ratio increased with duration of exposure from 0.79 in workers exposed 1-5 years to 2.79 in workers exposed >20 years, although the standardized mortality ratios did not reach significance (CI not reported). Lead exposure in U.S. industries has declined over the years, and this has been hypothesized as an explanation for the reduction in mortality from renal disease observed in this type of study. However, that fact that improved treatments for chronic renal disease have led to a decrease in mortality from end-stage renal disease (U.S. Renal Data System, 2004) may also be a factor. The mortality studies by Steenland et al. (1992) and others are further described in Annex Table AX6-4.4.

6.4.5 Lead Nephrotoxicity in Children

6.4.5.1 Studies in Adults Following Childhood Lead Poisoning

Henderson clearly established an increased risk for lead nephropathy in adult survivors of untreated childhood lead poisoning (Henderson, 1955). Lead nephropathy was responsible for substantial mortality in the Queensland, Australia population. However, as noted in the 1986 Lead AQCD, other studies of adults who survived childhood lead poisoning have not reported this degree of renal pathology. Studies published since 1986 have not observed the degree of renal pathology noted in the Queensland work either but have revealed some interesting findings. These studies, along with other studies of renal effects of lead in children, are presented in Annex Table AX6-4.5. Pertinent results are discussed below.

A study of comparing 21 adults, who had experienced childhood lead poisoning between 1930 and 1942, to age, sex, race, and neighborhood-matched controls found no significant differences in blood lead level, serum creatinine, or BUN (Hu, 1991). Mean measured creatinine clearance was unexpectedly higher in the previously lead-poisoned group compared to controls (112.8 versus 88.8 mL/min/1.73 m^2 [$p < 0.01$]). Mean in the lead-exposed group was also higher

than the predicted value of 94.2 mL/min/1.73 m^2 from the nomogram of Rowe et al. (1976). One survivor, who was identified but not included in the study, had been diagnosed with chronic interstitial nephritis on renal biopsy. Her blood lead was 30 μg/dL and her presentation was thus consistent with actual lead nephropathy. Strengths of this study included clear criteria for lead poisoning and assessment of clinical renal function that included both measured and estimated creatinine clearances. However, the study was limited by small size and the fact that the number enrolled was a very small subset of the initially identified cohort of 192. At least 43 (22.4%) of the 192 were confirmed to be deceased. That group had evidence of higher initial lead exposure, which raises concern regarding survivor bias in the study group. More importantly, the higher mean creatinine clearance in the lead exposed group provides further evidence for lead-related hyperfiltration. Again, as discussed in the occupational study section, this may hamper attempts to detect associations between lead dose and adverse renal effects.

Another study compared 62 participants who were diagnosed and chelated between 1966 and 1972 for initial blood lead levels >100 μg/dL to 19 age-matched siblings whose initial blood lead levels were <40 μg/dL (Moel and Sachs, 1992). Mean initial blood lead level was 150.3 μg/dL (SD 77.1) in the 62 survivors of lead poisoning; levels in the siblings could not be precisely quantified since values <40 μg/dL were not reported as exact values. Mean age at diagnosis was 2.2 years; age at follow-up was 22.2 years. Blood lead and renal function were serially monitored in the population. In 1983, mean blood lead in the poisoned group was statistically higher than in siblings (means of 14.5 and 11.6 μg/dL, respectively) but, by 1989, both groups had a mean lead of 7.4 μg/dL. Renal outcomes included serum creatinine, uric acid, and β_2-microglobulin, fractional excretion of β_2-microglobulin, urinary protein:creatinine ratio, and tubular reabsorption of phosphate. No statistically significant differences between previously lead-poisoned children and their siblings were observed. The prevalence of abnormal values between the two groups was not different. Initial blood lead level was not associated with serum creatinine, after adjustment for age, gender, and body mass index. However, since blood lead levels were not quantified in the siblings, their values were entered as 40 μg/dL in the continuous blood lead model. Lead was also entered as a dichotomous variable (poisoned versus siblings). Given the data available, the analysis was limited. No cumulative measure of lead was analyzed nor was the serially obtained data analyzed with longitudinal modeling. Four of 62 participants did experience an increase in serum creatinine such that their levels were

≥1.4 mg/dL by the end of the study period. These subjects may reflect lead-related renal toxicity, especially given the young age of the participants. However, it can be concluded that, despite lead exposures that are now considered extremely high, the degree of renal pathology was clearly not to the extent seen in Australia. The fact that these children were chelated when lead poisoning was diagnosed but the Queensland children were not may be an important distinction. Additional follow-up with longitudinal analysis would be of value in these children in order to evaluate their renal function as they develop other renal risk factors.

6.4.5.2 Lead Body Burden in Children with Chronic Renal Disease

Scharer et al. (1991) reported higher lead content in deciduous teeth in 22 German children, age 5-14 years, with varying degrees of renal insufficiency compared to a control group of 20 siblings or neighbors and a group of 16 children without known lead exposure. Mean dental lead content was 2.8, 1.7, and 1.4 μg/g, in the three groups, respectively. Lead levels in teeth were significantly higher in both the patient and sibling/neighbor control groups compared to the unexposed control group. Mean blood lead in the renal patients was only 2.9 μg/dL (range 1.1-10.1 μg/dL). Lead in teeth was not correlated with duration of renal impairment. However, reflective of the ongoing controversy at the time this article was published regarding whether decreased renal excretion causes increased lead storage, the authors attributed elevated lead levels to both exposure and accumulation from decreased renal excretion.

6.4.5.3 General Population Studies in Children

In the first study of the renal effects of lower level environmental lead exposure in children (as opposed to lead-poisoned children), Bernard et al. (1995) carried out a cross-sectional study of 195 children in the Czech Republic. One hundred forty-four children (63 boys and 81 girls) lived in 2 areas close to a lead smelter (designated as exposed groups one and two). A control group of 51 children lived in a rural area. Blood lead levels and urinary renal early biologic effect markers (RBP, β_2-microglobulin, NAG, albumin, and Clara cell protein) were obtained. Age ranged from 12 to15 years. In girls, mean blood lead concentrations in controls and exposure groups one and two were 8.4, 9.4, and 12.9 μg/dL, respectively. Corresponding values in boys were 8.7, 10.9, and 14.9 μg/dL, respectively. These levels were significantly

higher in both exposed groups compared to the control group. In contrast, blood cadmium levels were similar among all groups. After adjustment for age, sex, blood cadmium, and zinc protoporphyrin, log transformed blood lead was associated with log transformed RBP ($\beta = 0.302$, $p = 0.005$).

Verbeck et al. (1996) studied 151 Romanian children residing at various distances from a lead smelter. Associations between blood lead levels and renal outcome measures (urinary RBP, NAG, α_1-microglobulin, albumin, and alanine aminopeptidase) were analyzed. Mean age was 4.6 years (range of 3-6 years); gender was evenly divided. Mean blood lead was 34.2 µg/dL (SD 22.4), which is much higher than in Bernard et al. (1995). After adjustment for age and gender, a 10 µg/dL increase in blood lead was associated with a 13.5% (90% CI: 10.2, 17) increase in NAG excretion. No threshold was observed. Blood cadmium levels measured in a subset of the population were all <2 µg/L; however, this variable was not entered into the regression model.

Another study included 200 French children who resided close to smelters along with 200 age and gender matched controls recruited from areas believed to be unpolluted by heavy metals (De Burbure et al., 2003). Blood lead and cadmium were measured. Renal outcomes included urinary total protein, albumin, transferrin, β_2-microglobulin, RBP, brush border antigens, and NAG. Age ranged from 8.5 to 12.3 years. Geometric mean blood leads ranged from 2.7 µg/dL (SD 0.2) in female controls to 4.2 µg/dL (SD 0.2) in exposed males. The highest geometric mean blood cadmium was 0.52 µg/L. After adjustments for covariates, blood lead was not associated with any renal outcomes; however, blood cadmium was positively associated with NAG. This association was present in both control and exposed areas. Notably, the blood lead levels in this study were much lower than in the two studies discussed above.

Staessen et al. (2001) studied 200 17-year-old Belgian children. The two exposed groups were recruited from industrialized suburbs, whereas, the control group was recruited from a rural area. Mean blood lead levels were 1.5, 1.8, and 2.7 µg/dL in controls, and exposed groups one and two, respectively. The renal outcome measures analyzed were urinary β_2-microglobulin and serum cystatin-C. Although blood lead levels were low, after adjustment for sex and smoking status, blood lead was associated with both β_2-microglobulin and cystatin-C. Interestingly, blood cadmium was not associated with either outcome.

The current lack of sensitive markers of early renal damage that have been shown to predict subsequent renal function decline in longitudinal studies of lead exposed populations is problematic for research in this field. This is particularly true when studying children who do not have many of the other renal risk factors, such as hypertension and diabetes that older adults do. Coratelli et al. (1988) reported a decline in urinary NAG in association with a 1 month period of decreased occupational exposure in 20 adult lead battery factory workers followed over a 1 year period. Clinical renal function measures were not studied however. Sarasua et al. (2003) studied 526 adults and children, a mean of 4.5 years after an initial evaluation of renal function including measurement of urinary albumin, NAG, RBP, and alanine aminopeptidase. These participants were drawn from three populations exposed to volatile organic compounds and explosives via groundwater and controls. Follow-up was performed to determine if the EBE markers remained elevated and whether the presence of elevated EBE markers at baseline was associated with abnormalities in serum creatinine, serum cystatin C, 24-h creatinine clearance, and urine osmolality at follow-up. Among children who had elevated EBE markers at baseline, renal EBE markers remained elevated in 38%. However, none remained elevated in the 32 who had completed adolescence by the time of the follow-up. The authors noted the potential for puberty related biomarker changes. Further, abnormalities in the clinical measures were rare at follow-up. In contrast, elevated EBE markers at baseline in adults with chronic medical conditions of risk to the kidney, such as diabetes and to a lesser extent heart disease and hypertension, had persistent elevation of EBE markers and evidence of worse renal function at follow-up. Limitations of this study include limited data analysis, some loss to follow-up, and limited information on whether the original exposures for which these populations were studied may have influenced these results. The authors stated that no significant differences in renal outcomes between participants from exposed and control communities at follow-up were observed and noted that kidney function was not worse in exposed communities in the initial evaluations, which appeared to refer to the biomarkers as well. However, this report does not indicate whether exposure changed between the initial biomarker collection and the follow-up (i.e., decreased substantially or stopped altogether). Still the study illustrates the need for further prospective research to validate EBE markers in nephrotoxicant-exposed populations.

6.4.6 Mechanisms for Lead Nephrotoxicity

Individuals who have been heavily exposed to lead are at increased risk for both gout and renal disease (Shadick et al. 2000; Batuman 1993). Lead is thought to increase serum uric acid (urate) by decreasing its renal excretion (Emmerson, 1965; Ball and Sorensen, 1969; Emmerson and Ravenscroft, 1975). As discussed above, research in the last decade indicates that lead is nephrotoxic at lower levels than previously recognized. The same is true for uric acid (Johnson et al., 2003). Therefore, it is possible that one mechanism for lead-related nephrotoxicity, even at current lower levels of lead exposure, is via increasing serum uric acid.

In order to address this question, Weaver et al. (2005a) analyzed data from 803 current and former lead workers to determine whether lead dose was associated with uric acid and whether previously reported associations between lead dose and renal outcomes (Weaver et al., 2003) were altered after adjustment for uric acid. Outcomes included uric acid, blood urea nitrogen, serum creatinine, measured and calculated creatinine clearances, and urinary NAG and RBP. Mean uric acid, tibia lead, and blood lead levels were 4.8 mg/dL (SD 1.2), 37.2 μg/g bone mineral (SD 40.4), and 32.0 μg/dL (SD 15.0), respectively. None of the lead measures (tibia, blood, and DMSA-chelatable lead) were associated with uric acid, after adjustment for age, gender, body mass index, and alcohol use. However, when effect modification by age on these relations was examined, both blood and tibia lead were significantly associated in participants in the oldest age tertile ($\beta = 0.0111$ [95% CI: 0.003, 0.019] and $\beta = 0.0036$ [95% CI: 0.0001, 0.007]) for blood and tibia lead, respectively). These models were further adjusted for blood pressure and renal function. Hypertension and renal dysfunction are known to increase uric acid. However, they are also risks associated with lead exposure. Therefore, adjustment for these variables in models of associations between lead dose and uric acid likely results in overcontrol. On the other hand, since non-lead-related factors contribute to both renal dysfunction and elevated blood pressure, lack of adjustment likely results in residual confounding. Therefore, as expected, associations between lead dose and uric acid decreased after adjustment for systolic blood pressure and serum creatinine, although blood lead remained borderline significantly associated ($\beta = 0.0071$ [95% CI: −0.001, 0.015]). However, when the population was restricted to the oldest tertile of workers with serum creatinine greater than the median (0.86 mg/dL), likely the highest risk segment of the population, blood lead remained significantly associated with uric acid even after adjustment for systolic blood pressure and serum creatinine ($\beta = 0.0156$).

Next, in models of renal function in all workers, uric acid was significantly associated with all renal outcomes except NAG. Finally, in the oldest tertile of workers, after adjustment for uric acid, associations between lead dose and NAG were unchanged, but fewer of the previously significant ($p \leq 0.05$) associations noted between lead dose and the clinical renal outcomes in Weaver et al. (2003a) remained significant.

Data from the Normative Aging Study indicate that lead dose, at levels lower than those known to increase the risk for gout or in the study of Weaver et al. (2005a), is associated with increased uric acid (Shadick et al., 2000). Mean blood, patella, and tibia lead levels were 5.9 μg/dL, 30.2 μg/g bone mineral, and 20.8 μg/g bone mineral, respectively, in 777 participants. A significant association between patella lead and uric acid ($\beta = 0.007$ [[95% CI: 0.001, 0.013]; $p = 0.02$) was found, after adjustment for age, BMI, diastolic blood pressure, alcohol ingestion, and serum creatinine. Borderline significant associations between tibia ($p = 0.06$) and blood lead ($p = 0.1$) and uric acid were also observed. Notably these associations were significant even after adjustment for blood pressure and renal function, providing further evidence that low-level lead increases uric acid.

These data suggest that older workers comprise a susceptible population for increased uric acid due to occupational lead exposure. Uric acid may be one mechanism for lead-related nephrotoxicity. However, this is not the only mechanism, since in Weaver et al. (2005a), the association between blood lead and serum creatinine remained significant even after adjustment for uric acid. These mechanistic relations have more than just theoretical importance. Clinically relevant therapies may be possible since, as noted above, EDTA chelation has been reported to improve both renal function and urate clearance in patients with renal insufficiency and gout, even when EDTA-chelatable lead body burdens were low (Lin et al., 2001).

6.4.7 Susceptible Populations for Lead Nephrotoxicity

6.4.7.1 Chronic Medical Diseases

The general population studies by Tsaih et al. (2004) and Muntner et al. (2003) (discussed in section 6.4.4.1 General Population Studies above) indicate that patient populations with diabetes and hypertension are at increased risk for adverse renal effects of lead. The work of Lin and colleagues (several articles discussed in section 6.4.4.3 Patient Population Studies above) indicates that patients with CRI and gout are also at increased risk. In these settings, lead

appears to acts as a cofactor with other renal risk factors to cause early onset of renal insufficiency and/or a steeper rate of renal function decline. It is likely that the presence of larger high risk populations within general populations is an important factor in the lower lead dose thresholds noted for the adverse effects of lead on the kidney in environmental compared to occupational research.

6.4.7.2 Age

The work of Weaver and colleagues (discussed in Section 6.4.4.2.3 Korean Lead Workers above) suggests that older age is a risk factor. This is consistent with research in general populations (Lindeman et al., 1985) and is biologically plausible, since most renal risk factors increase with age. Gonick and Behari (2002) have summarized the data regarding the potential contribution of lead exposure to essential hypertension; similar issues may be involved with the renal dysfunction observed in aging.

6.4.7.3 Genetic Polymorphisms

6.4.7.3.1 δ-Aminolevulinic Acid Dehydratase (ALAD)

Research in the last two decades suggests that several genetic polymorphisms affect lead toxicokinetics (i.e., modify the relation between lead exposure and dose). Of those that are potentially relevant to the kidney, data on the gene that encodes for δ-aminolevulinic acid dehydratase (ALAD) are the most important in this regard. The ALAD enzyme is a principal lead binding protein; the isozymes in those with the ALAD2 allele are more electronegative and bind a greater proportion of blood lead than does the protein in individuals with the ALAD11 genotype (Bergdahl et al., 1997). Research to date indicates that individuals with the ALAD2 allele generally have higher blood lead levels than those with the ALAD11 genotype, although this may not be the case at lower levels of lead exposure (i.e., mean blood lead levels <10 μg/dL) (Kelada et al., 2001). Participants with the ALAD2 allele have been found to have lower bone lead levels in some studies (Hu et al., 2001; Kamel et al., 2003); other toxicokinetic differences have also been reported (Fleming et al., 1998; Hu et al., 2001; Schwartz et al., 1997; Smith et al., 1995). Overall, these data suggest that tighter binding of lead by the isozymes of the ALAD2 allele decreases lead sequestration in bone.

In contrast, data to determine whether the ALAD polymorphism impacts the renal toxicity of lead are still quite limited. The only environmentally exposed population in which this has been addressed is the Normative Aging Study. Wu et al. (2003) (discussed in detail in section 6.4.4.1.2 above) analyzed data to determine whether the ALAD genetic polymorphism modified associations between lead dose and uric acid, serum creatinine, and estimated creatinine clearance, 114 (16%) of the study group were either homozygous or heterozygous for the variant ALAD2 allele. None of the three outcomes were significantly different by genotype. However, effect modification by genotype on the association between tibia lead and serum creatinine was observed; the β coefficient (and slope) was greater in the group with the variant allele ($\beta = 0.002$ [SE not provided]; $p = 0.03$). Effect modification of borderline significance ($p < 0.1$) on relations between of patella and tibia lead with uric acid was observed; this was significant in participants whose patella lead levels were above 15 μg/g bone mineral ($\beta = 0.016$ [SE not provided]; $p = 0.04$). Similar to the serum creatinine model, patella lead was associated with higher uric acid in those with the variant allele. Genotype did not modify lead associations in models of estimated creatinine clearance.

The impact of the ALAD polymorphism on renal outcomes has been studied in four occupationally-exposed populations to date. The two that assessed both associations and effect modification by genotype are discussed here. Weaver et al. (2003b) analyzed data from 798 lead workers. Lead and renal function measures, as well as mean lead levels, were described in Weaver et al. (2003a) in Section 6.4.4.2 above. A total of 79 (9.9%) participants were heterozygous for the ALAD2 allele (none was homozygous). After adjustment, participants with the ALAD2 allele had lower mean serum creatinine and higher calculated creatinine clearance. Effect modification by ALAD on associations between blood lead and/or DMSA-chelatable lead and three of six renal outcomes was observed. Among those with the ALAD12 genotype, higher lead measures were associated with lower BUN and serum creatinine and higher calculated creatinine clearance. Among older workers (age ≥ median of 40.6 years), ALAD genotype modified associations between lead dose and uric acid levels. Higher lead dose was significantly associated with higher uric acid in workers with the ALAD11 genotype; associations were in the opposite direction in participants with the variant ALAD12 genotype (Weaver et al., 2005c).

Ye and colleagues (2003) assessed effect modification by ALAD on associations between blood lead with urinary NAG and albumin in a study of 216 lead workers. Geometric mean

blood lead was 37.8 µg/dL in 14 workers with the ALAD12 genotype and 32.4 µg/dL in workers with the ALAD11 genotype. After adjustment for age, NAG was borderline statistically higher in those with the variant allele whose blood lead levels were ≥40 µg/dL. In all lead workers, after adjustment for age, gender, smoking, and alcohol ingestion, a statistically significant positive association between blood lead and creatinine adjusted NAG was observed in the workers with the ALAD12 genotype but not in lead workers with the ALAD11 genotype (the groups were analyzed separately rather than in an interaction model).

Thus, two of the three studies reported steeper slopes for one or more associations between lead dose and adverse renal function in participants with the ALAD2 allele compared to those with the ALAD11 genotype which suggests that the variant ALAD gene confers additional risk for adverse renal outcomes in lead exposed populations. If the associations of Weaver et al., (2003b) represent lead-induced hyperfiltration their results could be consistent with increased risk from the variant allele as well. Ultimately, analysis of longitudinal data in the Korean lead worker population will be required to understand these complex relations.

6.4.7.3.2 BsmI Polymorphism of the Vitamin D Receptor (VDR) Gene

In contrast to ALAD, relatively few data on the *Bsm*I polymorphism of the gene that encodes for the vitamin D receptor (VDR) are available in lead exposed populations. Polymorphisms of the VDR gene are of interest in these populations due to the role of vitamin D and its receptor in regulating both intestinal calcium absorption and bone mineralization. These pathways impact lead absorption from the gastrointestinal tract and may impact lead storage and/or release from bone (Onalaja and Claudio, 2000). Analysis of data from the first evaluation of the Korean lead worker cohort found that participants with the B allele had significantly higher blood, DMSA-chelatable, and tibia lead levels than those with the bb genotype (Schwartz et al., 2000a); significantly higher patella lead in workers with the B allele was reported in data from the third evaluation (Theppeang et al., 2004). A study of 216 lead workers similarly reported higher blood lead levels in workers with the B allele (n = 20), after adjustment for age, gender, smoking, alcohol ingestion, and calcium ingestion, education, ALAD genotype, and ambient lead exposure (Ye et al., 2003). In a study of 504 former organolead manufacturing workers, with an average of almost two decades since last occupational exposure, VDR genotype was not associated with tibia lead concentrations (Schwartz et al., 2000c). However, the slope

of the positive association between age and tibia lead concentration was steeper in participants with the B allele compared to those with the bb genotype and tibia lead declined with years since last exposure in participants with the bb genotype, but increased in those with the B allele. In contrast, Chuang and colleagues (2004) found no difference in current or cumulative blood lead by *Bsm*I polymorphism in 544 lead workers.

Work in two of the populations described above has also provided information on the impact of the *Bsm*I VDR polymorphism. Of the 798 participants in Weaver et al. (2003b), 89 (11.2%) had genotype Bb or BB. No significant differences were seen in renal outcomes by VDR genotype nor was consistent effect modification observed. However, those authors assessed effect modification by this polymorphism on associations between patella lead and renal outcomes in current and former Korean lead workers in data from the third evaluation where patella lead was measured. Results were compared to those with three other lead biomarkers. The same six renal outcomes as in Weaver et al. (2003a) were measured. Mean blood, patella, tibia, and DMSA-chelatable lead were 30.9 μg/dL (SD 16.7), 75.1 μg lead/g bone mineral (SD 101.1), 33.6 μg lead/g bone mineral (SD 43.4), and 0.63 μg lead/mg creatinine (SD 0.75), respectively, in 647 lead workers (Weaver et al. [2005b]). Little evidence of effect modification by genotype on associations between patella lead and renal outcomes was observed. However, the VDR polymorphism did modify associations between the other lead biomarkers and serum creatinine and calculated creatinine clearance. Higher lead dose was associated with worse renal function in participants with the variant B allele. Models in two groups, dichotomized by median age, showed this effect was present in the younger half of the population. The authors were able to exclude different participant subsets as an explanation for the difference in VDR findings between the two evaluations. Longitudinal changes in renal function between evaluations may account for these findings and are currently being evaluated in longitudinal data analysis.

Ye and colleagues (2003) reported higher systolic blood pressure, after adjustment for age, in those with the variant allele whose blood lead levels were ≥40 μg/dL. In all lead workers, after adjustment for age, gender, smoking, and body mass index, a statistically significant positive association between blood lead and systolic blood pressure was observed in the 20 lead workers with the variant B allele, but not in lead workers with the bb genotype. Again, the fact that the genotype groups were analyzed separately, rather than in an interaction

model, decreased the study's power to detect a difference. This could be an explanatory factor for the lack of effect modification by VDR genotype on associations between blood lead and urinary albumin and NAG observed.

In conclusion, an increasing body of literature indicates that both of these polymorphisms affect lead toxicokinetics. However, data to determine if these polymorphisms impact renal function are still quite limited. Existing data are suggestive of an increased renal risk in lead exposed populations with the variant alleles of both polymorphisms.

6.4.8 Confounding of the Renal Effects of Lead by Other Potential Risk Factors

Studies selected for discussion in Section 6.4 above have generally controlled for at least the most basic risk factors known to affect renal function such as age, gender, and body mass index (or weight and height separately). Some have controlled for many other potentially important risk factors. In addition, exposure to other nephrotoxicants must be considered. Notably, although these are listed under confounders, some may be effect modifiers as well.

6.4.8.1 Cadmium

Similar to lead, cadmium is an ubiquitous nephrotoxicant that accumulates in the body. Environmental exposure in the United States occurs primarily through food and smoking (Agency for Toxic Substances and Disease Registry, 1993). Cadmium in food is a result of soil pollution from a variety of human activities such as phosphate fertilizer use, industrial releases from smelting, and fuel combustion. An analysis of NHANES III data, collected in a representative sample of the U.S. population from 1988-1994, indicates that mean urinary cadmium is 0.48 μg/g creatinine and 97.7% of the population has a level ≤2 μg/g creatinine (Paschal et al., 2000). Also similar to lead, cadmium causes proximal tubule pathology and is a risk factor for CRI.

The existing data indicate that cadmium, at exposure levels common in the U.S., confounds associations between lead exposure and at least one renal outcome, NAG. Roels et al. (1994) reported higher mean NAG in their lead-exposed group; however, NAG was correlated with urinary cadmium but not blood or tibia lead, despite the fact that mean urinary cadmium was only 1.04 and 0.53 μg/g creatinine in workers and controls, respectively. Cardenas et al.

(1993) reported a similar finding. Bernard et al. (1995) found an association between urinary cadmium and the NAG-B isoenzyme (released with breakdown of proximal tubular cells) in 49 cadmium workers and 20 age-matched controls. In multiple linear regression, urinary cadmium, but not lead, was associated with NAG-B, after adjustment for age. The association was significant even in the 44 participants with levels <2 μg/g creatinine. However, NAG-A (released by exocytosis) was correlated with urinary lead (the only lead measure), but not cadmium. Roels et al. (1995) reviewed data pertinent to the potential for cadmium confounding of associations between lead and NAG. In more recent work, Weaver et al. (2003a) measured urinary cadmium in a subset of 191 of the 803 workers in their study (mean urinary cadmium was 1.1 μg/g creatinine). Higher urinary cadmium levels were associated with higher NAG. Of the lead measures obtained, only tibia lead was significantly associated with NAG in the cadmium subset. When urinary cadmium and tibia lead were entered as covariates in the same model, both remained associated with NAG ($p < 0.05$). However, in comparing the effects, a 0.5 μg/g creatinine increase in cadmium had the same effect on NAG as a 66.9 μg/g bone mineral increase in tibia lead. When compared by ranges of exposure in this population, environmental level cadmium dose had a larger impact on NAG than did occupational lead dose.

Cadmium exposure may confound relations between lead exposure and other renal outcomes as well, although the data are too limited to draw firm conclusions. Positive associations between urinary cadmium, which is thought to be the best measure of cumulative cadmium exposure in the absence of cadmium-related renal damage, and low molecular weight (LMW) proteinuria are well established in the occupational setting. LMW proteinuria, most commonly assessed by β_2-microglobulin, is generally progressive at levels >1,500 μg/g creatinine in workers with substantial body burdens (one or more historical urinary cadmium >20 μg/g creatinine) but may also be progressive at lower levels (Roels et al., 1997; Bernard, 2004). More importantly, clinical renal function also declines as evidenced by decreasing GFR in cadmium exposed workers followed longitudinally after removal from exposure due to LMW proteinuria (Roels et al., 1989; 1997).

In contrast to the clear evidence that cadmium is a renal toxicant at occupational levels of exposure, the renal risk from lower level cadmium exposure remains uncertain. Most studies of environmental cadmium exposure are cross-sectional and have assessed EBE markers, rather than clinical renal outcomes (Alfven et al., 2002; Jarup et al., 2000; Noonan et al., 2002; Olsson

et al., 2002). The Cadmibel study, a general population study of exposed residents from both cadmium polluted and unpolluted areas (discussed in Section 6.4.4.1.1 above), found correlations between urinary cadmium and several urinary EBE markers (NAG, RBP, β_2-microglobulin, calcium, and amino acids) (Buchet et al., 1990). In those models, after adjustment for urinary cadmium and other covariates, blood lead was significant in models of β_2-microglobulin and amino acids but not NAG. However, in this same population, blood lead was inversely associated with creatinine clearance, whereas urinary and blood cadmium were not (Staessen et al., 1992). A 5 year follow-up was conducted to determine the significance of the EBE abnormalities (Hotz et al., 1999). In this study, models of renal function (two dichotomized outcomes: a 20% decline in creatinine clearance and a 20% increase in albumin excretion) in relation to quartiles of urinary cadmium and the EBE markers at baseline were analyzed by likelihood ratios. Baseline variables did not predict adverse renal outcomes. However, 25% of the original population was lost to follow-up; available data indicated that their baseline renal function was worse than those who participated in the follow-up study. This may have biased the study towards the null.

Two recent publications suggest that low-level cadmium exposure is associated with adverse clinical renal outcomes. Elevated urine cadmium levels were associated with prevalent microalbuminuria and decreased calculated creatinine clearance after adjustment for age, sex, race, smoking, and use of diuretics in an analysis of 16,094 participants in the NHANES III study (Young et al., 2004). Hellstrom et al. (2001) reported increased rates of renal dialysis and transplantation in residents of cadmium-polluted areas in Sweden. Compared to the "no-exposure group" (domicile >10 km from a battery plant), age-standardized rate ratios were 1.4 (95% CI: 0.8, 2.0) in the low-exposure group (domicile 2 to 10 km) and 1.9 (95% CI: 1.3, 2.5) in the moderate-exposure group (domicile <2 km). Exposure categorization was based on environmental monitoring in the study areas. Cadmium dose was not directly measured although occupationally exposed participants were considered in a separate group. Neither of these studies assessed lead exposure as a covariate, which would be important given the Cadmibel results (Staessen et al., 1992).

In conclusion, cadmium clearly confounds associations between lead dose and NAG. Given the similarities in both nephrotoxicants, cadmium may confound and/or modify associations between lead and other renal outcomes. However, data on the

concentration-response relation between environmental cadmium and the kidney are too limited to assess the potential for this at present. Future studies assessing both lead and cadmium are needed.

6.4.9 Summary of the Epidemiologic Evidence for the Renal Effects of Lead

In the last two decades, the quality of research on the renal impact of lead exposure has advanced dramatically. As a result, a much more accurate assessment of the adverse renal impact of lead exposure can now be made. Studies of environmentally-exposed general populations are one of the most important advances in this regard. The landmark Cadmibel study (Staessen et al., 1992) was the first to raise concern that the lead dose threshold for adverse renal effects in the general population was much lower than previously appreciated based on occupational data. Research in the Normative Aging Study population reached similar conclusions and suggested that both cumulative and circulating lead are associated with longitudinal decline in renal function. Diabetics were a particularly susceptible risk group in this regard. The NHANES III data analysis (Muntner et al., 2003) are notable for a sample size that is, by far, the largest of the environmental studies, comprehensive adjustment for other renal risk factors and the fact that population is representative of the U.S. population. Thus, the fact that renal dysfunction was observed in hypertensives at a mean blood lead of only 4.2 μg/dL and in the 1st quartile compared to the reference group (blood lead range from 2.5 to 3.8 μg/dL), provides strong evidence that the kidney is a target organ for adverse effects from lead at current U.S. environmental exposure levels.

Studies involving the longitudinal assessment of renal function decline in susceptible patient populations in relation to baseline chelatable lead body burden and therapeutic chelation constitute the other major advance in lead-renal research in the last two decades. Chelation was beneficial in an EDTA-chelatable lead level range from 80 or 150 to 600 μg/72 h, depending on the study (Lin et al., 1999, 2003). These studies suggested that lead body burden, at much lower levels than previously recognized, contributes to renal dysfunction in populations with CRI from a range of causes. This work also suggests that renal function in patients with CRI stabilizes and, in some cases, improves after therapeutic EDTA chelation of lead levels well below the level currently thought to require chelation.

A finding of note from the occupational studies is the observation of inverse associations (higher lead dose with lower BUN, serum creatinine, and/or higher creatinine clearance) in several studies. This may indicate lead-related hyperfiltration and have mechanistic implications. Regardless, significant associations could be obscured if opposite direction associations are present in different segments of the study population and interaction models to address this are not performed. This is a valid concern, since the settings in which these inverse associations are most likely are not well defined.

The renal impact in children from lead exposure at current environmental levels is difficult to assess, since the studies have involved measurement of EBE markers and their prognostic value is uncertain. Susceptible populations due to chronic medical diseases have clearly been established; risk from genetic polymorphisms may be important, but further study is required. Studies of potential mechanisms for the adverse renal effects of lead in humans, such as via uric acid have more than just theoretical importance, since EDTA chelation has been reported to improve both renal function and urate clearance in patients with renal insufficiency and gout (Lin et al., 2001). With an improved understanding of mechanisms, clinically relevant therapies may be possible.

6.5 CARDIOVASCULAR EFFECTS OF LEAD

6.5.1 Summary of Key Findings of the Cardiovascular Effects of Lead from the 1985 Lead AQCD and Addendum, and 1990 Supplement

The greater part of the evidence reviewed up to 1990 included analyses of the largest datasets available at the time, the National Health and Nutrition Evaluation Survey II (NHANES II), studying the U.S. population between 1976 and 1980, and the British Regional Heart Study (BRHS), studying men aged 40-59 years from 24 British towns. Analyses of the Welsh Heart Programme, a regional Welsh study, and the Caerphilly Collaborative Heart Disease Study, a cohort study of men aged 45-59 years living in one town in Wales, as well as smaller population and occupational exposure studies in the U.S., Canada, and Europe provided supporting evidence. These studies set enduring design and analysis standards by example for evaluating cardiovascular effects associated with blood lead levels in samples from diverse populations.

In general, the reviewed studies used multiple linear regression modeling of blood pressure and multiple logistic regression modeling of hypertension, cardiovascular mortality, and other cardiovascular disease, allowing adjustment of the blood lead effect on outcome by other factors known or suspected to be related to the exposure and outcome under study. The most commonly considered potential confounding factors were age, body mass index (BMI), alcohol use, and cigarette smoking.

These studies were almost exclusively cross-sectional studies, measuring cardiovascular outcome, blood lead, and control variables once, though one Canadian occupational study and one Danish birth-year cohort study used a longitudinal design. Studies sometimes presented analyses stratified by sex or age, by both sex and age, or by "race." Other analyses only reported results for one particular stratum. Separate analyses of datasets partitioned by stratified variables always reduce sample size available for statistical models, and, thereby, may reduce power to detect real effects.

Evaluated as a whole, the blood pressure studies supported a small but significant association between increasing blood lead concentrations and increasing blood pressure in study groups. The effect was more consistent across studies in middle-aged men than in other groups, ranging from a 1.5 to 3.0 mm Hg increase in systolic blood pressure for each doubling of blood lead from the mean blood lead level, and from a 1.0 to 2.0 mm Hg increase in diastolic blood pressure for each blood lead doubling, across a wide range of blood lead concentration down to 7 μg/dL. Most studies using multiple regression analyses stratified by sex were unable to find significant associations between blood pressure and blood lead in females, though one reanalysis of the NHANES II dataset did report a statistically significant relationship between diastolic blood pressure and lead in women aged 20 to 74 years. In studies reporting the use of different blood lead-blood pressure concentration-response relationships, log blood lead terms had lower probability values than linear blood lead terms, suggesting that increases in blood pressure with incremental fixed incremental increases in blood lead might be greater at lower blood lead concentrations than at higher concentrations.

Three studies of groups with occupational exposure reported mixed results. One study found significant excess mortality due to cardiovascular disease during the period 1946-1965 in a case-control study in the United Kingdom, but not 1966-1985. A study of U.S. battery and lead production workers from 1947-1980 found significant excess mortality due to "other

hypertensive disease" (codes 444-447 in the ICD 1955 classification system), but not due to hypertensive diseases outside those classifications. No excess mortality due to hypertension was found in a study of U.S. smelter workers between 1940 and 1965.

The BRHS study failed to reveal significant associations between blood lead and ischemic heart disease and stroke. However, electrocardiogram abnormalities associated with left ventricular hypertrophy were found related to blood lead in a subset of the NHANES II data, confirming an earlier study finding significant associations between ischemic changes and blood lead in lead workers.

Noninvasive measurement of bone lead concentration using XRF techniques was still maturing during the literature review period covered by the 1986 AQCD document and later addendum and supplement. No studies were reported using bone lead as a marker for lead exposure.

6.5.2 Effects of Lead on Blood Pressure and Hypertension

6.5.2.1 Introduction

Blood lead concentration remained the most widely used exposure index in blood pressure/hypertension epidemiologic studies from 1990 to present. Obtaining the sample is relatively noninvasive and quick, measurement techniques are well standardized and inexpensive with access to external quality assurance programs worldwide, and existing regulation and medical decision-making are based on blood lead levels. If exogenous lead exposure were the only determinant for blood lead concentration, it could be fair to state that a single blood lead measurement represented exposure to lead during the 30-90 day period preceding the measurement. However, blood lead concentration represents a combination of recent exposure to external sources and the influence of internal sources, principally bone lead. As detailed in Section 6.2, bone is a long-term storage depot for much of the lead absorbed by the body from external sources, and by weight can represent over 95% of the total body burden of lead in middle-aged persons, especially if external exposures are currently low. Bone lead has residence times of years to decades. Bones constantly absorb lead from and release lead to the circulatory system. Consequently, blood lead concentration is not only determined by current and recent past external exposure but is also influenced by existing bone lead concentration to a degree determined by current external exposure, accumulated past exposure stored in bones, and the

physiological state of the bones due to aging, disease, pregnancy, and lactation, among others. Studies using only blood lead concentration, as an exposure index cannot determine the relative contributions of current exogenous exposure and endogenous exposure to blood lead. Thus, they are unable to assess what part of measured blood lead effect on the circulatory system is due to possibly higher long duration past exposure and what part is due to the possibly immediate toxic effects of currently circulating lead. They are, instead, assessing a combined effect of past and present exposure in a proportion that will differ among subjects according to their past and present exposure, health history, and age.

Elevated blood pressure can be evaluated as a continuous measure (mm Hg) or as a dichotomized measure (hypertension). The definition of hypertension involves a categorical cut point of mm Hg above which one is hypertensive and below normotensive. Kannel (2000a,b) notes that this number has dropped over time for systolic/diastolic pressure and further notes a continuous graded influence of blood pressure even within what is regarded as the normotensive range. Some concern for an arbitrary definition as the cut point and one that has changed over time is a consideration. However defined, even if greater than the cut point at that time used clinically, the separation into these two groups offers a different perspective than blood pressure per se. Hypertension has a different clinical relevance than blood pressure changes themselves. The disease condition as an outcome and a change in mm Hg in relation to exposure both offer the opportunity for insight into the clinical relevance of the relationships. Biomarkers like bone lead and blood lead also add input into the acute/chronic nature.

The recently developed in vivo technique of XRF measurement of bone lead concentration has been used in a handful of studies to better assess the role of past exposure to lead on blood pressure and hypertension in essentially cross-sectional studies. Bone lead concentration provides a record of cumulative past exposure due to the long residence times of lead in bones, though the specific temporal pattern of past exposure cannot be readily determined from the measurement. Primarily cortical bones such as tibia have residence times measured in decades, whereas primarily trabecular bones such as calcaneus and patella have residence times measured in years to decades, reflecting different metabolic rates of the two bone types. As there is continual interchange of lead in bone and lead in blood, studies combining the measurement and modeling of both bone lead and blood lead have the best chance of dissecting out the roles of past and present lead exposure on blood pressure and hypertension.

The growing field of toxicogenetics now includes lead exposure epidemiology. The several studies combining subject evaluation of polymorphisms of genes thought to play a role in either the origin of cardiovascular disease, the toxicokinetics of lead or both are also reviewed. All epidemiologic studies of the cardiovascular effects of lead reviewed in this section as well as other additional studies are further summarized in Annex Table 6-5.1.

6.5.2.2 Blood Pressure and Hypertension Studies Using Blood Lead as Exposure Index

6.5.2.2.1 NHANES Studies

NHANES contributed the largest datasets analyzed in this review. As the surveys are also representative of the U.S. population, their results may be more readily applied to the general U.S. population than smaller cohort or occupational studies. The several papers using this dataset sometimes come to different conclusions, depending on the statistical techniques used in analysis, including logarithmic or linear specification of the lead variable, stratification of analyses according to sex or ethnic groups or use of interaction terms to define these groups, use of survey-design corrected models, choice of covariates in the models, and different age ranges analyzed.

NHANES II (1976-1980)

In one NHANES II-based study, males and females (number unreported but less than 9,000 combined) aged 20 to 74 years were studied with separate stepwise multiple regression models adjusted for sampling design (Schwartz, 1991). Mean blood lead levels were not reported. Covariates common to both male and female models were age and age^2, BMI, race, family history, cholesterol, zinc, tricep fold, and natural log lead. Models for men also included height and cigarette smoking. Natural log blood lead was significantly associated with diastolic blood pressure (systolic not reported) in males, with a 2.03 mm Hg diastolic (95% CI: 0.67, 3.39) increase for every doubling of blood lead, and for females a 1.14 mm Hg increase (95% CI: 0.13, 2.08). Interactions between blood lead and sex and between blood lead and race in a combined model were insignificant (not shown). The conclusion from these interaction terms is that the association between blood lead and diastolic blood pressure was not significantly different between men and women or between races. There was no mention of consideration of

model diagnostics, and the stepwise modeling may incorrectly include or exclude potentially confounding variables.

The other NHANES II-based study focused on black-white differences in blood pressure related to blood lead (Sorel et al., 1991). There were 473 blacks and 3,627 whites in the study, each nearly evenly divided by sex, aged 18 to 74 years. As is usual in U.S.-based studies, race/ethnicity was based on self-report. Survey design-adjusted multiple regression models were stratified on sex and included age, BMI, and linear blood lead as covariates. The effect of race and poverty index was assessed by including their terms in models with and without blood lead and determining change in race or poverty coefficients by comparing confidence intervals. Each 1 μg/dL increase in linear blood lead significantly predicted increased systolic blood pressure for both males (2.23 mm Hg [95% CI: 0.69, 3.61]), but not females (0.98 mm Hg [95% CI: −9.78, 3.06]) for each doubling of blood lead. The differences in black and white (race variable) blood pressure coefficients did not significantly change when lead was in or out of the model, either for subjects below the poverty index or above the poverty index. Race does not appear to significantly modify the relationship between blood lead and systolic blood pressure. The paper reported no model diagnostics. There were reporting inconsistencies in the female-stratified models, in which the coefficients and 95% CI did not correspond.

NHANES III (1988-1994)

A study using the NHANES III dataset from all adults 20 years of age and up examined the effect of natural log blood lead on systolic and diastolic blood pressure (Den Hond et al., 2002). Multiple regression analyses for each blood pressure measurement were stratified by sex and race, yielding four models for each blood pressure measurement. The mean blood levels were 3.6 μg/dL in white males (n = 4,685), 2.1 μg/dL in white females (n = 5,138), 4.2 μg/dL in black males (n = 1,761), and 2.3 μg/dL in black females (n = 2, 197). One group of covariates (age, age-squared, BMI, hematocrit, smoking, alcohol consumption, and an indicator variable for use of antihypertensive medications) were first entered as a block regardless of significance in each model, then another group of variables (coffee consumption, dietary calcium, dietary sodium/potassium ratio, total serum protein, total serum calcium, diabetes, and poverty index) was entered stepwise in the model without lead and the variable retained only if it was statistically significant ($p < 0.05$). Then log-transformed blood lead was forced into each model.

The model building procedure resulted in eight distinct models, each with their own unique mix of covariates. No model diagnostics were reported, nor was adjustment of results by survey sample weights and design. Only blacks had significant lead-systolic blood pressure associations; each doubling in blood lead was associated with a 0.90 mm Hg (95% CI: 0.04, 1.8) and 1.20 mm Hg (95% CI: 0.4, 2.0) increase in males and females respectively. The association of lead-diastolic blood pressure was also significant for black females (0.50 mm Hg [95% CI: 0.01, 1.1]). Interestingly, increasing blood lead was associated with significantly decreased diastolic blood pressure in white males (−0.6 mm Hg [95% CI: −0.9, −0.3]). The authors did not comment on their finding that the significant total serum calcium covariate in these two groups had opposite signs too (white male serum calcium $\beta = 6.50$ mm Hg/mmol/L, black female serum calcium $\beta = -5.58$ mm Hg/mmol/L). Though the authors offered no formal test of the difference between the two serum calcium coefficients, since both were significantly different than the null hypothesis coefficient of 0 and different in sign, it could be concluded that those coefficients were significantly different between the two groups. As the authors do not present the serum calcium coefficients before forcing lead into the models, it is not certain that blood lead in the model was associated with the significant sign difference of the calcium coefficients or if the calcium coefficients had opposite signs between the two groups without lead in the model. As each model had a different set of covariates, the presence or absence of one of the other covariates could have produced the same results. Nevertheless, this pattern of results may indicate significant confounding between serum calcium and blood lead associations with blood pressure. Though the study suggested differences between blacks and whites in response to lead, no statistical tests were performed of differences in lead coefficients based on race. In addition, the black-white effect differences associated with blood lead may be due to possible confounding in some or all of the models.

Limiting the study sample from NHANES III to women aged 40 to 59 years, another group of researchers addressed the relationship between blood lead and both blood pressure (n = 1,786) and hypertension (n = 2,165) (Nash et al., 2003). Blood pressure models excluded women who reported being under treatment for hypertension. Separate blood pressure multiple regression models were presented for diastolic and systolic blood pressure, each with and without stratification for dichotomous premenopausal/postmenopausal status. One block of covariates was entered without regard to statistical significance (age, race/ethnicity, BMI, and

serum creatinine). Another block of covariates (education, poverty income ratio, alcohol use, and cigarette smoking status) was entered second but only retained if variables were significantly associated with blood pressure. Finally, linear blood lead was forced in last. Logistic regression models for hypertension used the same covariate entry scheme with and without stratification on the menopause variable, but using a blood lead quartile exposure variable. Despite the stated procedure for covariate selection, all models used the same set of covariates: linear (or quartile) lead, age, race/ethnicity, alcohol use, cigarette smoking status, BMI, and serum creatinine. All models were adjusted for survey weights and design. Linear lead was significantly associated with systolic blood pressure only in the entire study sample; each 1 μg/dL increase in blood lead was associated with a 0.32 mm Hg (95% CI: 0.01, 0.63) increase in blood pressure. No associations were observed in the menopause-stratified analyses. Linear lead also was significantly associated with diastolic blood pressure in the entire study sample (0.25 mm Hg [95% CI: 0.07, 0.43]). Odd ratios of diastolic hypertension (>90 mm Hg) in logistic regression models was significantly related to blood lead with an odds ratio of 4.26 (95% CI: 1.36, 12.99) comparing the 1st quartile blood lead group (0.5-1.6 μg/dL) to the 4th quartile blood lead group (4.0-31.1 μg/dL) in all women not taking antihypertensive medications. Further stratification produced occasional significant odds ratios for either diastolic or systolic hypertension. There were some differences in table and text reporting of results and an inconsistency between the SE and the p-values.

Another study using the NHANES III database was notable for its formal testing of race and sex differences in lead effect by interactions terms (Vupputuri et al., 2003). The study used 5,360 white men (mean blood lead 4.4 μg/dL), 2,104 black men (mean blood lead 5.4 μg/dL), 5,188 white women (mean blood lead 3.0 μg/dL), and 2,300 black women (mean blood lead 3.4 μg/dL). Multiple linear and logistic regression models of blood pressure and hypertension (systolic ≥140 mm Hg, diastolic ≥90 mm Hg, and/or taking antihypertensive medication), respectively, were adjusted for age, high school education, BMI, alcohol, leisure-time physical activity, and dietary intake of sodium, potassium, and total energy. The models used linear lead, except for one set of hypertension models with a cut point for "high" lead exposure at ≥5 μg/dL. Subjects taking antihypertensive medication (n = 2,496) were not included in linear regression models of blood pressure. Neither age nor blood lead range were reported, nor was the technique of selecting and entering covariates in multiple regression models. Only coefficients for linear

lead effect for each model were reported. Significant interactions in multivariate models were found between lead and race and between lead and sex, though these analyses were not shown. Only black men and women had significant linear lead-blood pressure effects in adjusted systolic (0.25 mm Hg [95% CI: 0.06, 0.44] for black men and 0.47 mm Hg [95% CI: 0.14, 0.80] for black women with each 1 μg/dL increase in blood lead) and diastolic blood pressure (0.19 mm Hg [95% CI: 0.02, 0.36] for black men and 0.32 mm Hg [95% CI: 0.11, 0.54] for black women). Linear blood lead association with hypertension was significant only in women. The odds ratios were 1.09 (95% CI: 1.04, 1.13) for white women and 1.10 (95% CI: 1.06, 1.16) for black women for each 1 μg/dL increase in blood lead. The authors presented insufficient detail to evaluate this pattern of results. The use of diagnostic testing was not mentioned.

European Population Studies

The Health Survey for England 1995 examined a representative sample of the English population living in private households and provided up to 2,563 men and 2,763 women with a mean age of 47.6 years in a study of blood lead-blood pressure relationships (Bost et al., 1999). Precise blood lead range was not given but was at least from less than 1.5 μg/dL to greater than 8.5 μg/dL. The study used stepwise multiple regressions modeling of diastolic and systolic blood pressure stratified by sex, with and without adjustment for alcohol, and with and without subjects on antihypertensive medications. Candidate covariates, selected from a larger pool, included age, alcohol use (heavy drinkers versus all other drinkers and nondrinkers), SES (manual classes versus non-manual classes), location of residence in country (northern resident versus non-northern resident), smoking, and common log blood lead. As nonsignificant variables did not remain in the models, each model contained a unique mix of covariates. A doubling in blood lead in men was associated with an increase in diastolic blood pressure of 1.07 mm Hg (95% CI: 0.37, 1.78) when alcohol consumption was not in the model and 0.88 mm Hg (95% CI: 0.13, 1.63) when alcohol consumption was in the model. Women had a significant response to lead only for diastolic blood pressure in the model without adjustment for alcohol and with subjects using antihypertensive medication. There were no significant effects of lead on systolic blood pressure in any model. The authors provided no statistical justification for stratified modeling nor did they test for significant differences in lead coefficients as a result of the stratification.

U.S. Cohort Studies

The Boston-based Normative Aging Study, part of a longitudinal study of male veterans, examined the effects of blood lead on blood pressure in 798 men, aged 45-93 years old, with blood lead between 0.5 and 35.0 μg/dL (Proctor et al., 1996). Using multiple regression modeling with forced entry of natural log lead and other covariates (age, age^2, BMI, dietary calcium, exercise, smoking, alcohol, heart rate, and hematocrit), the authors found a significant increase of only diastolic blood pressure (0.83 mm Hg [95% CI: 0.08, 1.52]) for each doubling of blood lead. Though the relationship between blood lead and systolic blood pressure was positive, it was not significant. Nearly half the blood lead measures were derived from frozen red blood cells collected previously (up to several years earlier) and corrected for hematocrit determined at the time blood pressure was measured. Possible errors in correction of these samples and the non-contemporaneous nature of the resulting blood lead concentrations may have compromised the results.

Cheng et al. (2001), using the same Normative Aging Study data and stepwise multiple regression, found a near-zero association between systolic blood pressure and linear blood lead (−0.03 mm Hg for each μg/dL increase in blood lead) in 519 men aged 48 to 93. The subjects selected for this analysis were all free of hypertension (systolic>160 mm Hg or diastolic>95 mm Hg). Differences in subject selection procedures and modeling techniques may have accounted in the different results between Cheng et al. and Proctor et al. They also reported on incidence of hypertension developing between 1991 and 1997 using Cox proportional hazards models. Controlling for age, age^2, BMI, and family history of hypertension, linear blood lead was not significantly associated with risk of developing hypertension (systolic >140 mm Hg or diastolic >90 mm Hg) in normotensives at the start of the period (rate ratio of 0.98 [95% CI: 0.91, 1.06]) for each 1 μg/dL increase in blood lead.

Gerr et al. (2002) similarly reported near-zero linear blood lead effects on blood pressure on a combined group of 19-29 year old males and females (n = 502), half of whom had lived around active lead smelters as children, using forced entry of all covariates. Among the covariates forced into the model was tibia lead concentration, expected to be significantly correlated with blood lead. This may have reduced or confounded the effects of blood lead.

Korrick et al. (1999) examined linear and natural log blood lead effect on hypertension, defined as self-reported or physician diagnosis of hypertension or systolic or diastolic

≥140/90 mm Hg, in 284 middle-aged women from the Nurse Health Study based in Boston. The association of hypertension and blood lead was nonsignificant.

Rothenberg et al. (1999) tested a group of 1,527 women, aged 15 to 42 years, in their third trimester of pregnancy, with blood lead ranging from 0.5 to 40.4 μg/dL. They stratified testing into immigrant (n = 1,188) and nonimmigrant (n = 439) groups. They used forced entry of all covariates in multiple regression models, including natural log lead, age, BMI, coffee, iron supplement, and job stress, and found lead-related significant increases in systolic (1.18 mm Hg [95% CI: 0.45, 1.91] for each doubling of blood lead) and diastolic (1.02 mm Hg [95% CI: 0.37, 1.34]) blood pressure only in immigrants. The small size of the nonimmigrant group may have reduced power to detect significant effects. In a follow-up of 668 women returning for postpartum testing (Rothenberg, et al., 2002), using multiple regression models with forced entry of natural log blood lead, tibia and calcaneus lead, age, BMI, parity, smoking, immigrant status, and education, the authors found a significant decreases in systolic (−1.05 mm Hg [95% CI: −1.96, −0.14]) and diastolic (−1.16 mm Hg [95% CI: −1.98, −0.35]) blood pressure associated with doubling in blood lead in the postpartum women. This subgroup of women had no significant blood lead effects in the third trimester. Although the covariate pattern was different from the larger prenatal study (Rothenberg et al., 1999), thorough testing of possible confounding, especially with the bone lead measures, revealed no significant change in blood lead effects. This study finding is similar to that reported by Den Hond et al. (2002) for white males. No significant effect of blood lead on prenatal or postpartum hypertension (≥140/90 mm Hg) was found.

Morris et al. (1990) recruited a group of 105 women and 145 men, aged 18-80 years, from a clinic specializing in non-drug hypertension treatment. Blood lead ranged from 5-40.5 μg/dL. Multiple regression was performed with forced entry of natural log lead, age, BMI, dietary calcium, "other nutrients," serum ionized calcium, and erythrocyte protoporphyrin. Only men were found to have lead-related significant increases in systolic (3.17 mm Hg [95% CI: −2.13, 8.48] for each doubling of blood lead) and diastolic (1.32 mm Hg [95% CI: −2.12, 4.75]) blood pressure. Small study size limits conclusions based on nonsignificant findings in women. Dietary calcium is associated with reduced blood lead in many studies and could be considered a confounder with blood lead. Erythrocyte protoporphyrin is a biomarker of lead exposure and correlates with blood lead over at least part of the blood range in study subjects. There was the

inclusion of at least two collinear variables, a high proportion of covariates to subjects, and possible subject selection bias.

European Cohort Studies

The Glostrup Population Study (Copenhagen) studied 1,009 men and women (all born in 1936) longitudinally from 1976 to 1987 (Møller and Kristensen, 1992). Blood lead ranged from 2 to 62 µg/dL, depending on the year and sex stratum studied, with mean concentration dropping by ~40% over the study period. They used multiple regression with forced entry of natural log lead, BMI, tobacco use, and physical activity. Strongest associations between a doubling of blood lead and blood pressure were found early in the study period. In 1976, a doubling of blood lead was associated with 3.42 mm Hg (95% CI: 1.25, 5.58) increase in systolic blood pressure and 2.95 mm Hg (95% CI: 1.08, 4.83) increase in diastolic blood pressure in women. For men in 1981, a doubling of blood lead was associated with an increase of 1.89 mm Hg (95% CI: 0.00, 3.78) in systolic blood pressure and 1.14 mm Hg (95% CI: −0.37, 2.65) in diastolic blood pressure. No formal longitudinal analyses were performed, only analyses stratified by year and sex and analyses relating change in lead and other covariates to change in blood pressure from one study period to the next. As the relative risk of mortality was associated with increasing blood lead over the study period (see below), the general reduction in lead-associated blood pressure increase over the study period may have been masked by lead-associated mortality.

The Europe New Risk Factor Project in Rome collected data from 1,319 males aged 55-75 years with blood lead between 4.0 and 44.2 µg/dL (Menditto et al., 1994). They reported significantly increased systolic (4.71 mm Hg [95% CI: 2.81, 6.61]) and diastolic (1.25 mm Hg [95% CI: 0.33, 2.16]) blood pressure associated with a doubling of blood lead.

The Cadmibel studies from Belgium specifically selected part of their study group from those living near nonferrous smelters. Staessen et al. (1993) reported on 827 men and 821 women, aged 20 to 88 years, with blood lead ranging from 2.7 to 84.9 µg/dL for men and 1.3 to 42.4 µg/dL for women. They forced natural log blood lead into stepwise multiple regression models stratified by sex. Covariates available for selection were age, age^2, BMI, pulse rate, log gamma-glutamlytranspeptidase, serum total calcium, log serum creatinine, urinary potassium, smoking, alcohol, contraceptive use, and menopause. Near-zero nonsignificant relationships were found between blood lead and blood pressure for systolic blood pressure for

women and diastolic blood pressure for men and women. They reported a significant decrease in men's systolic blood pressure with increasing blood lead (−1.1 mm Hg for a doubling of blood lead), similar to the relationship found by Den Hond et al. (2002) for white men and by Rothenberg et al. (2002) for postpartum women. Stepwise regression results in different covariate patterns for each stratum and capitalizes on chance significance due to repeated testing.

A follow-up of the Cadmibel study, the PheeCad study, evaluated 359 men and 369 women, aged 20 to 82 years (Staessen et al., 1996). Fifty-nine percent of the men had occupational exposure. They were measured two times, at baseline and at follow-up about 5 years later. Women's mean blood lead at baseline and follow-up was 6.6 µg/dL (range 3.3-24.50 and 4.8 µg/dL (range 1.7-11.8). Men's mean blood lead at baseline and follow-up was 11.4 µg/dL (range 5.6-28.8) and 7.7 µg/dL (range 3.7-20.1). Multiple regression models were stratified on sex and in women on menopausal status. Time-integrated blood pressure measurements were used. Each doubling of blood log lead was significantly associated with a 5.19 mm Hg (95% CI: 1.05, 9.34) increase in diastolic blood pressure in 187 pre- and perimenopausal women. None of the other strata showed significant blood lead-related effects. Using 24-h ambulatory blood pressure readings during the follow-up showed significant associations between natural log blood lead and diastolic blood pressure in the group of all 345 women (2.42 mm Hg [95% CI: 0.00, 4.84]). There were no significant lead effects on systolic blood pressure in women or all blood pressure in men. Change in blood pressure and change in covariates between baseline and follow-up were used to assess the effect of change of blood lead in longitudinal analyses, similar to Møller and Kristensen (1992) above.
No significant effects of change in blood lead on change in blood pressure were found. Due to stratification and resulting small groups, there may have been reduced power to detect significant effects of lead.

U.S. Occupational Studies

Glenn et al. (2003) was one of the few studies to use a prospective design and was the only study using statistical techniques designed for repeated measures. They studied 496 male workers from New Jersey with former organolead exposure. Using generalized estimating equations with baseline linear blood lead, age, BMI, antihypertensive medication, smoking, education, measurement technician, and number of years to follow-up measurement of blood

pressure (range 10 months-3.5 years), they found that every 1 μg/dL increase in baseline blood lead was associated with 1.13 mm Hg/year (95% CI: 0.25, 2.02) increase in blood pressure over the observation period.

Schwartz et al. (2000b) reported significant blood lead associations with 543 male former organolead workers. Stepwise backward multiple regression showed an increase of 2.3 mm Hg in systolic blood pressure for each doubling in blood lead. The association with diastolic blood pressure was not significant.

Sharp et al. (1990) studied 132 black bus drivers (blood lead range 3.1-20.9 μg/dL) and 117 nonblack bus drivers (blood lead range 2.0-14.7 μg/dL) in San Francisco, aged 30 to 60 years. They used natural log blood lead in multiple regression models and found for each doubling of blood lead an increase of 5.22 mm Hg (95% CI: 0.60, 9.84) in systolic blood pressure among blacks, 3.27 mm Hg (95% CI: 0.10, 6.44) in diastolic blood pressure among blacks, and −3.96 mm Hg (95% CI: −8.32, 0.42) in systolic blood pressure among nonblacks.

Sokas et al. (1997) reported a possible race interaction ($p = 0.09$) on systolic blood pressure with linear blood lead in 264 construction workers aged 18-79 years. Each 1 μg/dL increase in blood lead increased systolic blood pressure in blacks by 0.86 mm Hg more than in whites. Neither the black or white lead coefficients were significant.

European Occupational Studies

Maheswaran et al. (1993) reported on 809 male factory workers with blood lead between less than 21 to more than 50 μg/dL from Birmingham, England. Unfortunately, the inclusion of factors that were related to blood lead, including additional direct measure of lead exposure in addition to linear blood lead, years working in factory, and inclusion of zinc protoporphyrin, may have biased the blood lead effect and resulted in nonsignificant lead effects on blood pressure.

Telišman et al. (2004) also reported nonsignificant effects of natural log blood lead on blood pressure in 115 male industrial workers with blood lead between 9.9 and 69.9 μg/dL, but included erythrocyte protoporphyrin in models, a variable correlated with blood lead over much of the observed blood lead range. Coefficients were not given, as lead did not enter into stepwise regression models.

Asian Occupational Studies

Male and female factory workers (n = 798) from Chonan, Korea (blood lead between 17.8 and 64.8 µg/dL) were studied principally for the effects of genotype of ALAD and vitamin D receptor on cardiovascular response to lead (Lee et al., 2001). These aspects are covered more thoroughly below. As part of their work, the authors developed multiple regression models examining the effect of linear blood lead on blood pressure with forced entry of age and age^2, BMI, sex, antihypertensive medication, lifetime alcohol, and ALAD and vitamin D genotypes. A marginally significant effect of blood lead on systolic blood pressure (diastolic blood pressure not modeled) was noted, with a 10 µg/dL increase in blood lead associated with a 0.7 mm Hg (95% CI: −0.04, 1.4) increase in blood pressure.

Nomiyama et al. (2002) used a combined group of 193 female crystal glass workers and nonexposed controls, aged 16 to 58 years, with blood lead between 3.8 and 99.4 µg/dL. The authors used a stepwise multiple regression with a novel technique to reduce collinearity among covariates. From a large group of covariates, they selected covariates eligible to enter the regression from a factor analysis. Although the stepwise entry of these variables resulted in different models for systolic and diastolic blood pressure, both models included linear blood lead, age, urine protein, and plasma triglycerides. The diastolic model additionally included family hypertension and low density lipoprotein. Each 10 µg/dL increase in blood lead was significantly associated with a 1.26 mm Hg (95% CI: 0.58, 1.94) increase in systolic blood pressure and a 1.05 mm Hg (95% CI: 0.52, 1.57) in diastolic blood pressure. In alternative models with ordered categories of blood lead, systolic blood pressure was 7.5 mm Hg (95% CI: 3.0, 12.0) and diastolic blood pressure was 6.3 mm Hg (95% CI: 3.4, 9.1) higher in workers with blood lead ≥60 µg/dL than in controls with <11.4 µg/dL. Models did not control for BMI.

Wu et al. (1996) examined the effect of ordered blood lead category on blood pressure of 112 male (aged 18-67 years) and 110 female (aged 18-71 years) lead battery factory workers in multiple regression models. Blood lead ranged from 8.3 to 95.4 µg/dL. Nonsignificant blood lead effects were found possibly due to the inclusion of two additional lead exposure measurements, ambient air lead and work history, likely leading to substantial collinearity with blood lead.

Meta-Analysis of Blood Lead-Blood Pressure Studies

The most recent meta-analysis of the blood lead-blood pressure literature analyzed 31 studies from a large pool of studies published up to 2001 (Nawrot et al., 2002). Two other meta-analyses were also published during this reporting period (Schwartz, 1995; Staessen et al., 1994), covering many of the earlier papers cited in Nawrot et al. (2002), and derived similar coefficients for the lead effect, so they will not be reviewed here. The meta-analysis authors selected studies with 50 or more subjects, with subjects 10 years of age and up, with blood pressure and blood lead measurement techniques presented in sufficient detail to estimate effect sizes, and with preference given to papers with models adjusting for age, BMI, and "additional factors of proven importance." Where possible, studies with stratified analyses based on sex and race were entered in the meta-analysis as separate subgroups. Studies were weighted by the number of subjects to arrive at estimates and CIs for lead effect on diastolic and systolic blood pressure. Nearly half the studies reported lead effects from linear lead terms, the remainder from log-transformed lead. To include both types of studies in the analyses, the authors reported effect sizes based on doubling the mean blood lead concentration. For models using logarithmic blood lead, this doubling has the same effect anywhere in the range of blood lead in the study. For models using linear blood lead, the doubling effect was referenced from the mean blood lead reported. Figures 6-5.1 and 6-5.2 depict the effect estimates for systolic and diastolic blood pressure, respectively, included in the meta-analysis from Nawrot et al. (2002). Ninety-five percent CIs overlapped for males and females and for blacks and whites, indicating no significant differences in lead effect by gender or race. In the group of studies as a whole, the combined meta-analysis coefficients for each doubling of blood lead were highly significant for both systolic (1.0 mm Hg [95% CI: 0.5, 1.4]) and diastolic (0.6 mm Hg [95% CI: 0.4, 0.8]) blood pressure. The meta-analysis supports the statistical association between increased blood lead and increased blood pressure over a wide range of populations in many studies. Two major systematic reviews of the lead-blood pressure literature were published during this review period (Hertz-Picciotto and Croft, 1993; Research Triangle Institute, 1999). The findings here, especially noting the significant effects from the meta-analysis (Nawrot et al., 2002), continue to support the significant association between blood lead and blood pressure/hypertension in diverse segments of the general population.

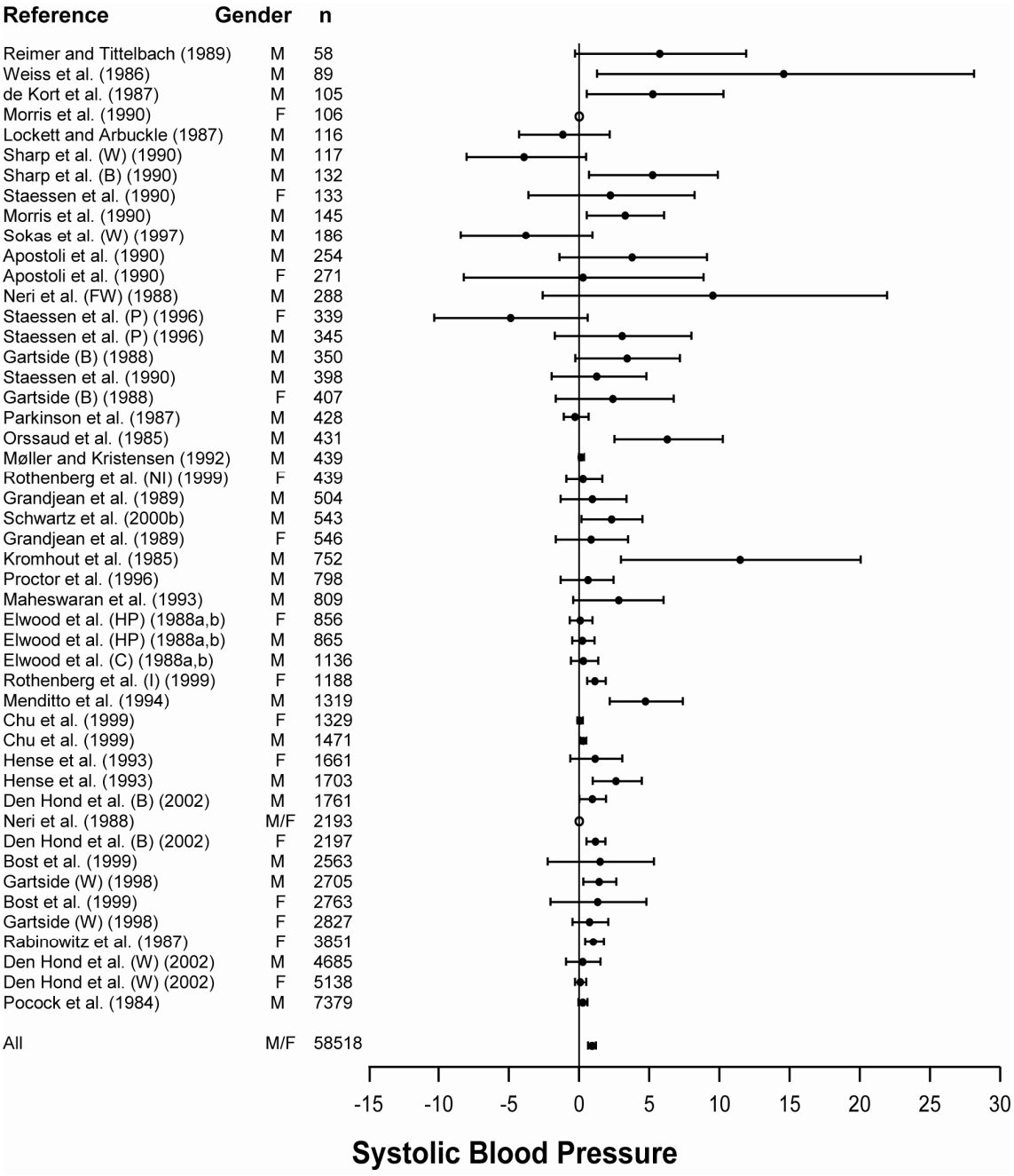

Figure 6-5.1. Change in the systolic pressure (effect estimate in mm Hg) associated with a doubling of the blood lead concentration. Studies arranged vertically by increasing study size.

Study key: C = Caerphilly Study, HP = Welsh Heart Program, P = PheeCad Study, W = whites, B = blacks, NI = nonimmigrants, I = immigrants, FW = foundry workers, CS = civil servants.

Source: Nawrot et al. (2002).

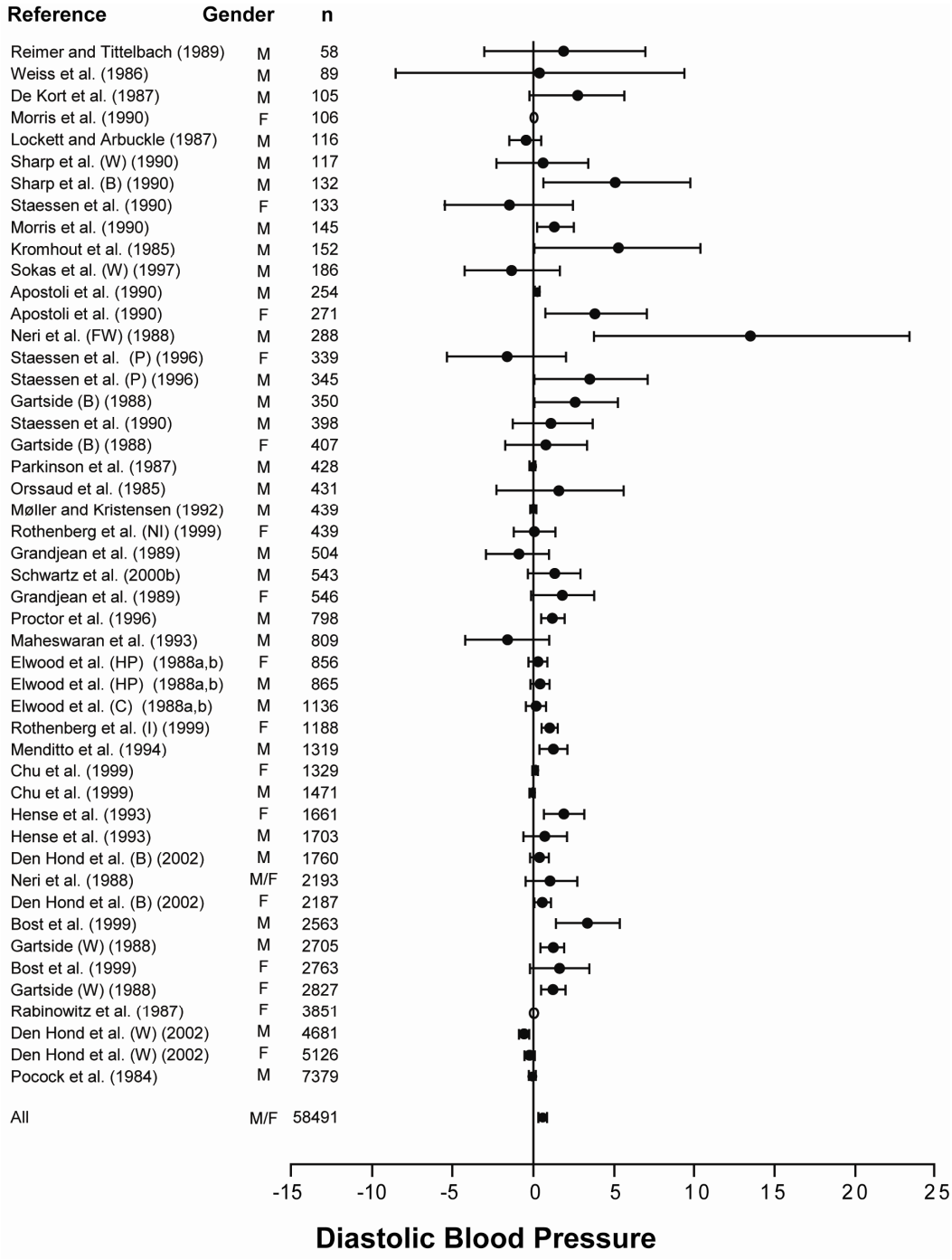

Figure 6-5.2. Change in the diastolic pressure (effect estimate in mm Hg) associated with a doubling of the blood lead concentration. Studies arranged vertically by increasing study size.

Study key: C = Caerphilly Study, HP = Welsh Heart Program, P = PheeCad Study, W = whites, B = blacks, NI = nonimmigrants, I = immigrants, FW = foundry workers, CS = civil servants.

Source: Nawrot et al. (2002).

Quantitative estimates for the effect of doubling the mean blood lead concentration on systolic and diastolic blood pressure from the various studies discussed here are summarized in Tables 6-5.1 and 6-5.2. Results from these individual studies also generally appear to agree with the results of the meta-analysis by Nawrot et al. that increased blood lead levels are significantly associated with increased systolic and diastolic blood pressure. Figures 6-5.1 and 6-5.2 graphically depict the results of many of the studies discussed. The effect estimates in the figures also reflect the effect of doubling blood lead on blood pressure. Most of the effects are based on concurrent blood lead. Effects for the entire study population are presented unless only effects in subsamples are reported. Other selection criteria used in a few instances were subjective.

A Bayesian meta-analysis was performed to examine the use of log-linear and linear blood lead in blood pressure regression models (Figure 6-5.3). A significant blood lead effect on systolic pressure was observed using both log-linear and linear blood lead. Heterogeneity also was significant for both log-linear lead ($p = 0.0002$) and linear lead ($p = 0.05$). The source of heterogeneity could be due to several factors, including different methods of statistical analysis, different study protocols, and subject selection differences. The log-linear and linear lead effects were 0.62 mm Hg (95% CI: 0.12, 1.11) and 0.98 mm Hg (95% CI: 0.51, 1.45), respectively, for systolic blood pressure. The difference between the effect estimates from using log-linear or linear lead was nonsignificant. This meta-analysis suggests there are significant differences between the studies, but overall there is a combined effect of blood lead on systolic blood pressure. The fact that several individual studies did not detect a significant effect may be due to small study size or other factors affecting effect measurement precision.

6.5.2.3 Blood Pressure and Hypertension Studies Using Bone Lead as Exposure Index

Korrick et al. (1999) used a case-control design to study the relationship between hypertension and three measures of lead exposure (blood lead, tibia [cortical bone] lead, and patella [trabecular bone] lead in women. The final study sample consisted of 89 hypertension cases and 195 controls, excluding those with history of hypertension, cardiovascular disease, renal disease, diabetes, or malignancy, use of antihypertensive medications, BMI $\geq$29, and incomplete data, aged from 47 to 74 years. Cases were selected through a randomization procedure that produced approximately equal numbers of cases for each of three blood pressure

Table 6-5.1. Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Linear Lead (Coefficients Represent Effect of Doubling Blood Lead Calculated from Mean Blood Lead or Mid-point of Range)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES										
Cheng et al. (2001) Boston, U.S. Normative Aging Study 1991-1997	Males (519) [mean age = 66.4]	5.9[a] (3.9)[d] [range not given]	Multiple linear regression forced entry of linear lead, age, age^2, BMI, family hypertension, smoking, alcohol, dietary calcium.	Not stated	Not applicable	Alcohol, smoking, age, dietary calcium	−0.2 (−2.0, 1.6) [0.92]	Not tested	Not tested	—
Hu et al. (1996)	—	—	Almost completely overlapping with Cheng et al. (2001).	—	—	—	—	—	—	—
Gerr et al. (2002) Eastern Washington, Western Idaho, U.S. 1994	Females and Males (502) [19-29]	2.2[a] (1.9)[d] [<1->7]	Linear multiple regression with forced entry of linear blood lead, tibia lead, age, BMI, sex, education, birth control pills, smoking, height, hemoglobin, serum albumin, childhood residence, income, alcohol.	Not stated	Not applicable	Age, tibia lead, smoking, alcohol, hemoglobin	0.0 (−1.1, 1.1) [0.57]	Not applicable	Age, tibia lead, smoking, alcohol, hemoglobin	−0.3 (−2.8, 1.6) [0.48]
Nash et al. (2003) U.S.-NHANES III 1988-1994	Females (1,786) [40-59]	2.9[a] (not stated)[d] [0.5-31.1]	Multiple linear regression (survey weighted), excluding treated hypertensives, with forced entry of linear lead, age, BMI, race/ethnicity, serum creatinine. Education, poverty, alcohol, smoking among variables for stepwise entry.	Not stated	Not applicable	Alcohol, smoking, age	0.9 (0.0, 1.8) [0.46]	Not applicable	Alcohol, smoking, age	0.7 (0.2, 1.2) [0.26]

Table 6-5.1 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Linear Lead (Coefficients Represent Effect of Doubling Blood Lead Calculated from Mean Blood Lead or Mid-point of Range)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)d or (IQR)e [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES (cont'd)										
Vupputuri et al. (2003) NHANES III, U.S.	Black female (2,300) [≥18-?]	3.4[a] (3.3)[d] [not stated]	Multiple linear regression (not survey corrected) with forced entry of all variables: linear lead, age, BMI, education, alcohol, physical activity, dietary Na, K, and calories.	Not stated	Not applicable	Age, alcohol	1.6 (0.5, 2.7) [0.82]	Not applicable	Age, alcohol	1.1 (0.4, 1.8) [0.36]
	Black male (2,104) [≥18-?]	5.4[a] (3.3)[d] [not stated]					1.3 (0.3, 2.4) [0.54]			1.0 (0.1, 2.0) [0.48]
	White female (5,188) [≥18-?]	3.0[a] (3.3)[d] [not stated]					0.3 (−0.4, 1.1) [0.38]			0.0 (−0.5, 0.4) [0.22]
	White male (5,360) ([≥18-?]	4.4[a] (3.3)[d] [not stated]					0.4 (−1.0, 1.8) [0.72]			0.0 (−0.5, 0.5) [0.27]
Sorel et al. (1991) NHANES II, U.S. 1976-1980	Female (2,056) [18-74]	13.2[a] (not stated) [not stated]	Multiple linear regression (survey corrected) with forced entry of linear lead, age, BMI.	Not stated	Not applicable	Age	Errors in original article	Not applicable	Age	Errors in original article
	Male (2,044) [18-74]	20.1[a] (not stated) [not stated]					1.7 (−0.6, 3.8) [0.56]			2.6 (0.8, 4.2) [0.87]

Table 6-5.1 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Linear Lead (Coefficients Represent Effect of Doubling Blood Lead Calculated from Mean Blood Lead or Mid-point of Range)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES (cont'd)										
Hense et al. (1994) Augsberg, Denmark 1987-1988	Female (no alcohol) (701) [28-67]	Not stated (not stated) [<3->8]	Multiple linear regression with forced entry of linear lead, age, BMI, hematocrit, residence, smoking in women. In men, analyses were stratified by place of residence.	Not stated	Not applicable	Age, smoking, hematocrit	0.4 (−2.6, 3.4) [1.53]	Not applicable	Age, smoking, hematocrit	1.2 (−0.8, 3.1) [0.99]
	Female (<40 g/day) (877) [28-67]	Not stated (not stated) [<3->10]					0.7 (−1.5, 2.9) [1.12]			1.5 (0.1, 2.9) [0.71]
	Female (≥40 g/day) (83) [28-67]	Not stated (not stated) [<3->14]					11.0 (3.8, 18.3) [3.64]			7.3 (2.8, 11.8) [1.36]
	Male, urban (118) [28-67]	Not stated (not stated) [not stated]	No blood lead values shown; thus, there are no coefficients calculated and shown				—			—
	Male, rural (no alcohol) (147) [28-67]	Not stated (not stated) [<5->11]	Multiple linear regression with forced entry of linear lead, age, BMI, hematocrit, residence, smoking in women. In men, analyses were stratified by place of residence.	Not stated	Not applicable	Age, smoking, hematocrit	2.8 (−2.4, 8.0) [2.63]	Not applicable	Age, smoking, hematocrit	0.2 (−3.3, 3.8) [1.80]
	Male, rural (<40 g/day) (463) [28-67]	Not stated (not stated) [<6->12]					5.8 (1.9, 9.8) [2.01]			3.5 (0.8, 6.2) [1.37]
	Male, rural (≥40 g/day) (356) [28-67]	Not stated (not stated) [<7->15]					5.0 (0.6, 10.9) [2.87]			3.3 (0.3, 6.3) [1.53]

Table 6-5.1 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Linear Lead (Coefficients Represent Effect of Doubling Blood Lead Calculated from Mean Blood Lead or Mid-point of Range)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
OCCUPATIONAL STUDIES										
Glenn et al. (2003) New Jersey, U.S. 1994-1998	Males, former organolead workers	—	Only measured change in blood pressure over a period ranging from 10 months to 5 years. Design not commensurate with other studies in this table. See Glenn (2001), below.	—	—	—	—	—	—	—
Glenn et al. (2001) New Jersey, U.S. 1996-1997	Males, former organolead workers (213) [mean age = 58.0]	5.2[a] (3.1)[d] [not stated]	Subset of Schwartz (2000), see below.	—	—	—	—	—	—	—
Schwartz et al. (2000b) New Jersey, U.S. 1996-1997	Males, former organolead workers (543) [41.7-73.3]	4.6[a] (2.6)[d] [1-20]	Multiple backward elimination stepwise linear regression models. Pool of available covariates not specified.	Not stated	Age, BMI, smoking, antihypertensive medications	Age, smoking	2.3 (0.2, 4.4) [1.15]	Age, BMI, smoking, antihypertensive medications	Age, smoking	1.3 (−0.3, 2.9) [0.86]
Sokas et al. (1997) Maryland U.S. 1989-1990	Male, construction workers (264) [18-79]	8.0[a] (not given) [2-30]	Multiple linear regression, presumably with forced entry (not stated). Covariates available for entry not stated. Insufficient information given for separate black and white blood pressure effects on blood pressure.	Not stated	Linear lead, BMI, age, hematocrit, erythrocyte protoporphyrin, race, race by lead interaction	Age, hematocrit, erythrocyte protoporphyrin	Not given here, as coefficient was altered by presence of race-lead interaction	Linear lead, BMI, age, hematocrit, erythrocyte protoporphyrin, race, race by lead interaction	Age, hematocrit, erythrocyte protoporphyrin	Not given here, as coefficient was altered by the race-lead interaction

Table 6-5.1 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Linear Lead (Coefficients Represent Effect of Doubling Blood Lead Calculated from Mean Blood Lead or Mid-point of Range)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
OCCUPATIONAL STUDIES (cont'd)										
Maheswaran et al. (1993) Birmingham Europe Dates not given	Male factory workers (809) [mean age 43.3]	31.5[b] (5.5)[d] [<21->50]	Multiple linear regression, forced entry of age, BMI, alcohol, zinc protoporphyrin, years working, smoking, linear lead.	Not stated	Not applicable	Age, alcohol, zinc protoporphyrin, years working, smoking	2.2 (−0.9, 5.4) [1.60]	Not applicable	Age, alcohol, zinc protoporphyrin, years working, smoking	−1.3 (−3.5, 0.9) [1.12]
Lee et al. (2001) Chonan, Korea 1997-1999	Male and female lead-using factory workers (798) [17.8-64.8]	32.0[a] (15.0)[d] [4-86]	Multiple linear regression with forced entry of age, age^2, sex, BMI, antihypertensive medication, lifetime alcohol, ALAD and vitamin D receptor genotypes.	Not stated	Not applicable	Age, alcohol	2.2 (−0.1, 5.5) [1.18]	Not shown	Not shown	—
Lustberg et al. (2004) Chonan, Korea 1997-1999	Overlapping study with Lee (2001), above	—	Used deciles as lead variable.	—	—	—	—	—	—	—
Nomiyama et al. (2002) Beijing, China Dates not given	Female crystal toy makers and nonexposed sewing workers (193) [16-58]	37.6[a] (9.2)[d] [3.8-99.4]	Multiple linear regression with forward stepwise addition of variables ($p < 0.2$). Candidate variables were selected from a very large group and narrowed down to ten by factor analysis. The ten available for entry were not stated.	Not stated	Linear lead, age, urine protein, plasma triglyceride	Age	4.7 (2.0, 7.4) [1.35]	Linear lead, age, urine protein, plasma triglyceride, family hypertension, low density lipoprotein	Age	3.9 (1.9, 5.9) [1.04]
Wu et al. (1996) Central Taiwan Data collection dates not given	—	—	Ordered categories of lead used. Coefficients cannot be calculated.	—	—	—	—	—	—	—

Table 6-5.2. Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Logarithmic Lead (Coefficients Represent Effect of Doubling Blood Lead)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES										
Den Hond et al. (2002) NHANES III U.S. 1988-1994	Black males (1,761) [≥20]	4.2[c] (2.7, 6.5)[e] [<1.2->20.0]	Multiple linear regression (no adjustment for survey design): forced entry of log lead, age, age^2, BMI, hematocrit, smoking, alcohol, antihypertensive drugs. Forward stepwise entry ($p < 0.05$) among coffee, dietary calcium, dietary Na/Ca, serum total protein, serum total calcium, diabetes.	Not stated	Serum total protein	Alcohol, smoking, age	0.9 (0.04, 1.8) [0.43]	Coffee, dietary Na, diabetes, serum total protein,	Age, smoking, alcohol, dietary calcium	0.3 (−03, 1.0) [0.16]
	Black females (2,197) [≥20]	2.3[c] (1.4, 3.9)[e] [<0.8->9.0]			Dietary Na/Ca, serum total protein	Alcohol, smoking, age	1.2 (0.4, 2.0) [0.42]	Dietary Na/K, serum total protein, serum total calcium	Age, smoking, alcohol	0.5 (0.01, 1.1) [0.26]
	White males (4,685) [≥20]	3.6[c] (2.3, 5.3)[e] [<1.2->15.0]			Dietary calcium, dietary Na/Ca, serum total protein, serum total calcium	Alcohol, smoking, age, dietary calcium	0.3 (−0.2, 0.7) [0.23]	Dietary Na/Ca, serum total protein, serum total calcium, diabetes	Age, smoking, alcohol	−0.6 (−0.9, −0.3) [0.15]
	White females (5,138) [≥20]	2.1[c] (1.3, 3.4)[e] [<0.8->8.0]			Serum total protein, diabetes	Alcohol, smoking, age	0.1 (−0.4, 0.5) [0.32]	Dietary calcium, serum total protein, diabetes	Age, smoking, alcohol, dietary calcium	−0.2 (−0.5, 0.1) [0.14]
Morris et al. (1990) Oregon, U.S. 1984-1989	Females (106) [18-80]	6.9[a] (3.6)[d] [?-39]	Linear multiple regression with stepwise entry among ln lead, age, BMI, dietary calcium, "other nutrients," serum Ca^{2+}, erythrocyte protoporphyrin.	Not stated	Age, dietary calcium, BMI	Age, dietary calcium	Not stated	Age, dietary calcium, hemoglobin	Age, dietary calcium	Not stated
	Males (145) [18-80]	8.0[a] (4.4)[d] [5-40.5]			Age, ionized calcium	Age	3.2 (Not given)	Age, hemoglobin, smoking	Age, hemoglobin, smoking	1.3 (Not given)

Table 6-5.2 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Logarithmic Lead (Coefficients Represent Effect of Doubling Blood Lead)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES (cont'd)										
Proctor et al. (1996) Normative Aging Study Boston, U.S. 1991-1993	Males (798) [45-93]	6.5a (4.0)d [0.5-35.0]	Multiple linear regression with forced entry of all variables: ln lead, age, age^2, BMI, dietary calcium, exercise, smoking, alcohol, heart rate, hematocrit.	Not stated	Not applicable	Alcohol, smoking, dietary calcium, age	0.6 (−0.8, 1.9) [0.69]	Not applicable	Alcohol, smoking, dietary calcium, age	0.8 (0.1, 1.5) [0.41]
Rothenberg et al. (1999) Los Angeles, U.S. 1995-1998	Pregnant women, immigrant (1,188) [>14-<44]	2.3b (1.3)d [0.5-40.4]	Multiple linear regression with forced entry of all variables: ln lead, age, BMI, coffee, iron supplement, job stress.	Residual analyses, outliers, heteroscedasticity	Not applicable	Age	1.2 (0.5, 1.9) [0.37]	Not applicable	Age	1.0 (0.4, 1.5) [0.28]
	Pregnant women, nonimmigrants (439) [>14-<44]	1.9b (+1.3, −0.8)[d] [not stated]				Age	0.3 (−1.1, 1.6) [0.67]		Age	0.1 (−1.3, 1.4) [0.67]
Rothenberg et al. (2002) Los Angeles, U.S. 1995-2001	Females third trimester (637) [15-43]	1.9b (+3.6, −1.0)[d] [not stated]	Multiple linear regression with forced entry of all variables: ln lead, age, BMI, education, immigrant status, smoking, alcohol, parity. Hypertensive subjects ≥140/90 mm Hg (third trimester or postpartum) were excluded.	Residual analyses, outliers, heteroscedasticity	Not applicable	Alcohol, smoking, age	0.0 (−0.9, 0.8) [0.43]	Not applicable	Alcohol, smoking, age	0.1 (−0.5, 0.8) [0.34]
	Females postpartum (637) [15-43]	2.3b (+4.2, −1.2)[d] [0.4-23.7)					−1.0 (−2.0, −0.1) [0.46]			−1.2 (−2.0, −0.3) [0.41]

Table 6-5.2 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Logarithmic Lead (Coefficients Represent Effect of Doubling Blood Lead)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
POPULATION OR COMMUNITY STUDIES (cont'd)										
Schwartz (1991) NHANES II U.S. 1976-1980	Females (<5,000) [20-74]	Not stated	Multiple linear regression with forced entry of ln lead, age, age^2, and BMI. Stepwise entry ($p < 0.05$) among race, cigarettes/day, tricep skinfold, family hypertension, exercise, ln serum Zn, dietary K and Na, serum cholesterol, height, and ln dietary vitamin C.	Not stated	No results presented	No results presented	No results presented	Race, zinc, family hypertension, tricep fold, cholesterol	Age	1.1 (0.2, 2.0) [0.48]
	Males (<5,000) [20-74]							Race, zinc, family hypertension, tricep fold, cholesterol, height, cigarettes	Age, cigarettes	2.0 (0.6, 3.4) [0.69]
Bost et al. (1999) Health Survey for England 1995	Females (2,763) [16-not given]	2.6[b] (not stated) [<1.5->8.5]	Multiple linear stepwise regression (forward or backward not stated), with possible entry of age, log BMI, log lead, alcohol, social class, place of residence, and smoking.	Not stated	Age, BMI, residency	Age	No results presented (nonsignificant)	Age, BMI, alcohol	Age, alcohol	No results presented ($p > 0.05$)
	Females (2,563) [16-not given]	3.7[b] (not stated) [<1.5->8.5]			Age, BMI, alcohol, social class	Age, alcohol	No results presented (nonsignificant)	Age, BMI, alcohol	Age, alcohol	0.9 (0.1, 1.6) [0.38]
Menditto et al. (1994) Rome, Europe New Risk Factor Project 1989-1990	Males (1,319) [55-75]	11.3[a] (not stated) [4.0-44.2]	Multiple linear regression with forward stepwise entry among ln lead, age, BMI, heart rate, lipids, triglycerides, glucose, smoking, alcohol, skinfold.	Not stated	BMI, heart rate, serum lipids, age, glucose, smoking, skinfold, triglycerides, skinfold	Smoking	3.9 no other measures given	BMI, heart rate, age, smoking, lipids, triglycerides	Smoking	1.2 no other measures given

Table 6-5.2 (cont'd). Systolic and Diastolic Blood Pressure and Blood Lead Modeled with Logarithmic Lead (Coefficients Represent Effect of Doubling Blood Lead)

Reference, Study Location, Period	Study Population, Sample Size (n), Age [years]	Blood Lead Mean[a], Geom. Mean[b], or Median[c] (SD)[d] or (IQR)[e] [range] µg/dL	Model Type and Variables Considered	Model Diagnostic	Systolic Blood Pressure (mm Hg)			Diastolic Blood Pressure (mm Hg)		
					Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]	Unique Covariates	Lead-Associated Variables	Coefficient (95% CI) [SE]
OCCUPATIONAL STUDIES										
Sharp et al. (1990) San Francisco, U.S. 1986	Male bus drivers, black (132) [30.4-60.7]	6.5[b] (+2.7, −1.9)[d] [3.1-20.9]	Linear multiple regression analysis of unspecified type. All models presented here adjusted for ln lead, age, age^2, BMI, caffeine, smoking. Also examined effect of stratified caffeine modeling (not shown here as group sizes were less than 100), and effect of successive addition of covariates on lead coefficient, and including alcohol use (not shown).	Used influence diagnostics to identify two influential subjects. Analyses stratified by race showed maximum 10% change in lead coefficients without influential subjects. Data shown here includes all subjects.	Not applicable	Age, smoking	5.2 (0.6, 9.8) [2.3]	Not applicable	Age, smoking	3.3 (0.1, 6.4) [1.6]
	Male bus drivers, nonblack (117) [30.6-58.9]	6.2[b] (+2.7, −1.8)[d] [2.0-14.7]					−3.9 (−8.3, 0.4) [2.3]			0.5 (−2.3, 3.4) [1.4]
Telišman et al. (2004) Zagreb, Croatia 2000-?	Male industrial workers (115) [20-43]	36.7[b] (not given) [9.9-69.9]	Linear multiple regression with forward stepwise entry among ln or linear lead, years of exposure, age, smoking, alcohol, BMI, ALAD, erythrocyte protoporphyrin, blood Cd, serum Zn, serum Cu. Neither form of lead variable was reported as significant, but coefficients not shown. Even though forward stepwise was used and insignificant lead variables were not shown, other nonsignificant variables were shown in models.	Not stated			None given			None given

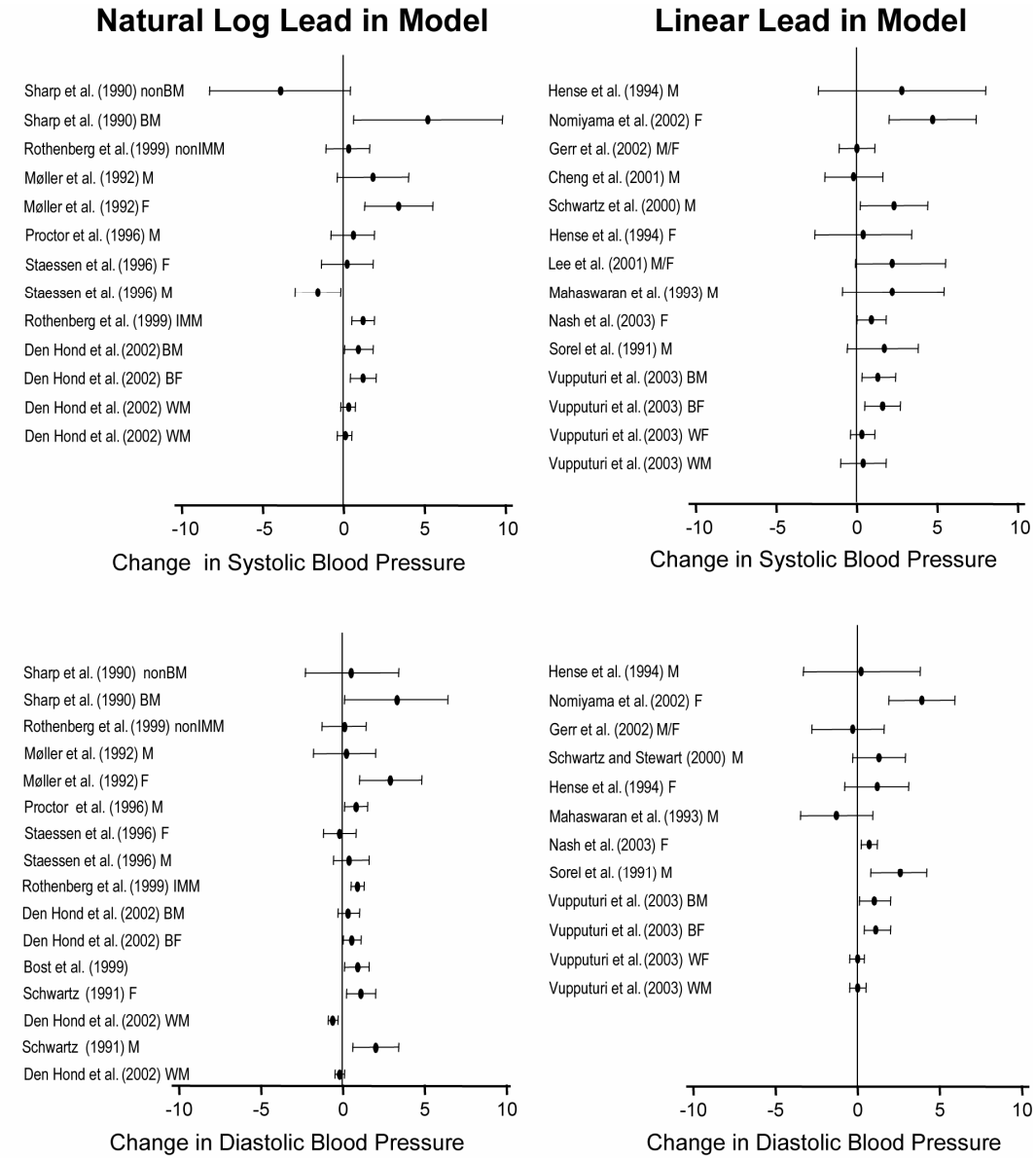

Figure 6-5.3. Effect of doubling mean blood lead on estimate of blood pressure change with 95% CIs. Studies arranged vertically by increasing study size. Where multiple models from the same study were presented, such as repeated measures over time or adding a confounding variable, only the effect estimate from the first model is shown. When the same study was multiply published with subsamples, only the effect estimate from largest study is shown.

Study key: B = blacks, W = whites, M = males, F = females, IMM = immigrants, non-IMM = nonimmigrants.

categories, hypertensive (≥140 mm Hg or 90 mm Hg), high normal (≥121/75 mm Hg up to hypertension limit), and low normal (<121/75 mm Hg). As many as four controls were matched to cases by 5 year age grouping. Though they did not match cases and controls on other potential confounding variables, they included these variables in their models. The dependent variable was constructed by placing blood pressure measurements into the three groups. The mean blood lead level was 3.1 μg/dL; the mean tibia and patella lead levels were 13.3 μg/g and 17.3 μg/g, respectively. An ordered logistic regression with proportional odds assumptions was used to asses linear blood lead, patella and tibia bone lead effects on odds of hypertension, controlling for age, BMI, dietary calcium, alcohol use, dietary sodium, smoking, and family hypertension. They presented results from four models with the same covariates determined a priori, but with each lead variable tested separately. Only patella lead concentration significantly ($p = 0.03$) predicted increased odds for hypertension, but the effect was small. Each 10 μg/g increase in patella lead was associated with an odds ratio of 1.28 (95% CI: 1.03, 1.60). Separate analyses testing interactions of alcohol use, age, and menopausal status showed no significant interaction with patella lead, though the small sample size had little power to detect significant interaction effects. Model diagnostics were given for justifying the use of proportional odds ordinal regression but none were given justifying use of a linear blood lead term in the models.

Rothenberg et al. (2002) investigated associations between both hypertension and blood pressure with blood lead, tibia lead, and calcaneus lead in 668 women, aged 15-44 years, in the third trimester of pregnancy and during a 3-month postpartum period using a cohort design and multiple logistic and multiple linear regression modeling. Subject exclusion criteria were blood lead > than 5 geometric SDs from the geometric mean, documented renal disease, cardiovascular disease, diabetes, use of stimulant drugs, and extreme postnatal obesity (BMI >40). Geometric mean prenatal and postnatal blood lead levels were 1.9 μg/dL and 2.3 μg/dL, respectively. Mean tibia and calcaneus lead levels were 8.0 μg/g and 10.7 μg/g, respectively. Variables in all models were selected a priori and retained in the models regardless of significance level. Control variables were education, smoking status, immigrant status, parity, age, and BMI in all models. Prenatal models also controlled for postpartum hypertension in lieu of family history of hypertension. None of the subjects used antihypertensive medications during the study. All three lead variables were simultaneously tested in all models. Third trimester blood lead

ranged from 0.4 to 30.0 μg/dL, postpartum blood lead ranged from 0.2 to 25.4 μg/dL. Calcaneus lead ranged from −30.6 to 49.9 μg/g and tibia lead ranged from −33.7 to 42.5 μg/g. Only calcaneus lead was significantly associated with an increase in hypertension (either ≥140 mm Hg systolic or ≥90 mm Hg diastolic) during pregnancy, with an odds ratio of 1.86 (95% CI: 1.04, 3.32) for each 10 μg/g increase of calcaneus lead. No association between calcaneus lead and hypotension was found postpartum. The authors found the same pattern of trabecular lead concentration association with blood pressure during but not after pregnancy in normotensive women. A 10 μg/g increase in calcaneus lead was associated with ~0.75 mm Hg (95% CI: 0.04, 1.46) increase in systolic and ~0.58 mm Hg (95% CI: 0.01, 1.16) increase in diastolic blood pressure in the third trimester. Only blood lead, tested simultaneously with tibia and calcaneus lead, was significantly associated with postpartum maternal blood pressure, but the relationship was negative, higher blood lead associated with lower postpartum blood pressure. For a doubling in blood lead, the systolic blood pressure increased −1.05 mm Hg (95% CI: −1.96, −0.14) and diastolic increased −1.16 mm Hg (95% CI: −1.98, −0.30). Though the authors thoroughly explored lead interaction with all other covariates in the models, they were unable to discover an effect modifier among them to explain the relationship. Postpartum physiological changes were discussed in relation to this last result. Thorough diagnostic testing was performed for all models. Only linear age terms were used in the models without exploration of age^2 terms. The authors did not use the repeated measures nature of the design in their analyses, instead they analyzed third trimester pregnancy data and postpartum data separately. They did not statistically test differences in coefficients from the same variables in the two parts of the study.

Two studies examined a subset of subjects participating in the Normative Aging Study. Hu et al. (1996) used a cross-sectional design of 590 men with median age in the mid-60s (range 48-92 years). Blood lead ranged from 1 to 28 μg/dL, tibia lead from <1 to 96 μg/g, and patella lead from 1 to 142 μg/g. Logistic regression models were initially constructed by adding age, race, BMI, family history of hypertension, smoking, alcohol use, and dietary sodium and calcium. Testing linear blood lead, tibia lead, and patella lead one by one against hypertension status (systolic >160 mm Hg, diastolic >96 mm Hg, or taking antihypertensive medication), they found no significant relationships with any of the lead variables, each entered separately. Only when they used backward elimination of nonsignificant variables did they find a significant odds ratio of 1.50 (95% CI: 1.09, 2.10) for each doubling of tibia lead from the mean (20.8 μg/g) for

hypertension. Later, Cheng et al. (2001) followed up the same group, constructing a multiple linear regression model for systolic blood pressure (diastolic blood pressure was not mentioned in model descriptions) in subjects not hypertensive at baseline measurement. They used a fixed set of control variables, including age and age terms, BMI, family history of hypertension, and alcohol and calcium intake, selected by univariate and bivariate testing of a larger set. After entering linear blood lead, tibia, and patella bone lead separately into the models, they reported a significant association only with tibia lead (1.60 mm Hg [95% CI: 0.00, 4.44] increase in systolic blood pressure for each doubling of tibia lead from the mean). Several years later (not specified in methods but no more than 6 years), the group of subjects that was originally not classified as having definite hypertension was retested for presence of definite hypertension (≥160/95 mm Hg). Each lead measure was separately entered into a Cox's proportional hazards model of incident definite hypertension. Only patella lead showed a significant increase in the rate ratio in subjects with no history of definite hypertension, 1.14 (95% CI: 1.02, 1.28) for each 10 μg/g increase in patella lead. Similar results were obtained when the borderline hypertensive group (>140/90 mm Hg) was combined with the definite hypertension group in patella lead. A rate ratio of 1.23 [95% CI: 1.03, 1.48]) was estimated. Use of linear lead terms may have affected the ability of the studies to detect significant blood lead effects.

A pair of studies using the same group of male workers (age range 42-74 years) previously exposed to organic and inorganic lead at an industrial plant in the U.S. investigated the role of blood lead and bone lead on blood pressure. Blood lead ranged between 1 and 20 μg/dL and tibia lead ranged from −1.6 to 52 μg/g. The study by Schwartz et al. (2000b) controlled for age, BMI, current smoking, and current use of antihypertensive medication in backward elimination linear multiple regression models for blood lead, tibia lead, and DMSA-chelatable lead, forcing each lead term into separate models. Only blood lead was a significant predictor of blood pressure. In multiple logistic regression models, only blood lead in workers <58 years of age was significant in predicting hypertension (>160/96 mm Hg). Although this study used linear blood lead in one model, it used another model with both linear and squared-blood lead. Both lead terms were significant in the respective models. In a follow-up study (Glenn et al., 2003) with most of the same subjects of the first study, subsequent measurements of blood pressure occurred at intervals of 4-12 months for 10.2 months to 3.5 years. The study was notable not only for its prospective nature but in the use of statistical models adjusting for

repeated measurements. Models were constructed by adding to a base model containing age at start of study, race, BMI, and indicator variables for technician. Lead variables were always forced in the models, but it is not clear if they were each tested separately. Other potential confounder variables were added stepwise to the model if they met a probability criterion. Both increasing linear blood lead and tibia lead were significantly associated with increasing systolic blood pressure times the number of years of follow-up blood measurement, but not with change in diastolic blood pressure. Each 10 μg/g increase in tibia lead was associated with a 0.78 mm Hg, year (95% CI: 0.24, 1.31) increase in systolic blood pressure for workers followed for the longest time. No model diagnostics were reported.

Gerr et al. (2002) tested the effect of blood lead and tibia lead only in young adults (age 19-29 years), both males and females, on blood pressure. Half the subjects had grown up around an active lead smelter. Multiple linear regression models always used age, sex, height, BMI, current smoking status, frequency of alcohol consumption, current use of birth-control medication, hemoglobin level, serum albumin, and income, regardless of significance levels. Both blood lead (as a linear term) and bone lead (a four category ordinal variable from <1 μg/g to >10 μg/g) were tested together. Tibia lead concentration in the highest group was associated with a significant increase in both systolic (4.26 mm Hg) and diastolic (2.80 mm Hg) blood pressure when compared to the lowest tibia lead group. No model diagnostics were presented.

6.5.3 Other Cardiovascular Outcomes

6.5.3.1 Ischemic Heart Disease

A community-based case-referent study taken from the Stockholm Heart Epidemiology Program compared survivors of first-time myocardial infarction with matched referents based on sex, age, year of study enrollment, and hospital catchment area (Gustavsson et al., 2001). The authors assessed lead exposure by a three category ordinal scale based on lead levels in airborne dust. In the comparison of unexposed to >0-0.03 mg/m^3 (mean 0.01 mg/^{3}) and unexposed to >0.04 mg/m^3 (mean 0.10 mg/m^3), the relative risk was 0.88 (95% CI: 0.69, 1.12) and 1.03 (95% CI: 0.64, 1.65), respectively.

In a reanalysis of the NHANES II dataset, the influence of linear blood lead in the diagnosis of left ventricular hypertrophy (LVH) based on examination of electrocardiograms and body habitus data in less than 9,900 subjects (exact number not given) of age 25-74 years was

tested in a survey-adjusted stepwise logistic regression model (Schwartz, 1991). The final model adjusted LVH by age, race, and sex. The odds ratio for LVH was 1.33 (95% CI: 1.20, 1.47) for each 10 µg/dL increase in blood lead over an unreported blood lead range. The author reported no significant interactions between blood lead and race or between blood lead and sex, though the article noted that the number of cases of LVH was small. The linear lead effect had greater significance than the natural log lead effect, the reverse of the relationship between the two lead specifications usually seen when blood pressure is the outcome variable.

In another study of electrocardiograms in 775 men (mean age 68 years, range 48-93) from the Normative Aging Study, patella and tibia lead concentrations were significantly associated with increased heart rate-corrected QT and QRS intervals in men under 65 years but not over 65 years in multiple regression stepwise analysis (Cheng et al., 1998b). Only tibia lead concentration was significantly associated with an increased odds ratio of intraventricular conduction deficit (2.23 [95% CI: 1.28, 3.90]) for every 10 µg/g increase in tibia lead), but only in men under 65 years. In contrast, both tibia and patella lead concentration was significantly associated with atrioventricular conduction deficit (odds ratio of 1.22 [95% CI: 1.02, 1.47] and 1.14 [95% CI: 1.00, 1.29] for each 10 µg/g increase in tibia and patella lead, respectively), but only for men greater than or equal to 65 years. None of the lead measurements were significantly associated with arrhythmia. Linear blood lead terms were not significantly associated with any of the above outcomes. Though the authors reported examining both saturated models (models with all considered control and confounding variables, significant or not) and stepwise models, only stepwise models were presented or discussed with each lead term forced into separate models. Thus, each model had an individual mix of control/confounding variables, though age was common to all models. Despite using age as a control/confounding variable in all models, the article offered no statistical justification for the age-stratified analysis.

A group of male and female battery factory workers (n = 108) working for at least 10 years and who were hired from 1960 to 1983 had blood lead levels from 1970 to 1994 ranging from 5 to 93 µg/dL (Tepper et al., 2001). Using a fixed covariate multiple logistic regression model, including age, BMI, sex, and family history of hypertension, the authors found a nonsignificant odds ratios for risk of hypertension (>165/96 mm Hg or self-reported use of hypertension medications) comparing the first tertile (138-504 µg/dL·year) cumulative blood lead index with the third tertile (747-1447 µg/dL·year) index. Echocardiogram left ventricular

mass was not significantly related to cumulative blood lead index or time-weighted average blood lead.

The discrepancy in blood lead results between the two electrocardiogram studies by Schwartz (1991) and Cheng et al. (1998b) could well be explained by population differences. Though both used large datasets, the age range of the NHANES II subject pool was between 25 and 74 years and used both men and women, whereas the age range for the Normative Aging study was 48 to 93 years and used only men. Furthermore, the Cheng et al. study had 775 subjects whereas the Schwartz had a much larger, though unspecified number. The Tepper et al. (2001) study had the least number of subjects (n = 108), which may have resulted in not detecting significant effects on a different measure of LVH. Nonetheless, the two electrocardiogram studies each reported a significant lead effect, and the study with bone lead (Cheng et al., 1998b) is particularly interesting, not only for its older sample but because the bone lead exposure measure reflected accumulated past exposure, which blood lead only partly reflects. The two studies are in agreement that lead exposure, either past or present, is significantly associated with ischemic heart disease.

6.5.3.2 Stroke

No published articles relating lead specifically to stroke were uncovered, though some ICD diagnostic codes (the cerebrovascular codes) reported in other parts of this section included stroke.

6.5.3.3 Cardiovascular/Circulatory Mortality

A recent follow-up of the NHANES II cohort provided mortality data used to associate past blood lead concentration with increased circulatory mortality in the U.S. population (Lustberg and Silbergeld, 2002). Blood lead concentration as measured during 1976-1980 was divided into three categories (<10 μg/dL, 10-19 μg/dL, and 20-29 μg/dL) after eliminating 109 subjects with blood lead ≥30 μg/dL, leaving 4,190 subjects 30-74 years of age in the mortality sample followed to the end of 1992. During the follow-up period, 929 subjects died of all causes. ICD-9 codes 390-459 (circulatory) accounted for 424 deaths. Proportional hazards models using a priori selected potential confounding variables (age, sex, race, education, income, smoking, BMI, exercise, and location) were used to calculate risk ratios of cardiovascular

mortality for the two higher lead categories compared against a <10 µg/dL reference. The 20-29 µg/dL category showed a significant relative risk of 1.39 (95% CI: 1.01, 1.91) for cardiovascular mortality.

Another longitudinal study combined fatal and nonfatal coronary heart disease (ICD-8 codes 410-414) and cardiovascular disease (ICD-8 codes 410-414 and 430-435) categories from a Danish 1936 birth cohort (N = 1052) followed from 1976-1990 (Møller and Kristensen, 1992). During the study period, 54 cases of cardiovascular disease with 19 deaths were reported. Log-transformed blood lead was used in a Cox proportional hazards model, controlling for a priori selected variables of tobacco use, cholesterol, physical activity, sex, systolic blood pressure, and alcohol. Two other models were also examined, those leaving out alcohol or both alcohol and systolic blood pressure. None of the adjusted models showed significant risk hazard for combined fatal and nonfatal cardiovascular disease, though blood lead was significantly associated with outcome in all models except the one containing both alcohol and systolic blood pressure for "total mortality" risk hazard, which presumably counted noncardiovascular mortality as well (not detailed in article). This article is notable for its detailed discussion of using confounding variables, such as hemoglobin and alcohol use, in multivariate models of lead-cardiovascular associations. Small sample size and low death rate may have contributed to the nonsignificant results.

An occupational study, using 1,990 male workers who worked at least 1 day between 1940 and 1965 in an active lead smelter in the U.S. (mean length of employment at smelter 13.8 years; mean estimated length of lead exposure 9.9 years), failed to show an association with lead and standardized mortality ratios compared to the U.S. population reference group up to 1988 (Steenland et al., 1992). Neither mortality from ischemic heart disease (ICD-9 410-414), hypertension with heart disease (ICD-9 402 and 404), hypertension with no heart disease (ICD-9 401, 403, and 405), nor cerebrovascular disease (ICD-9 430-438) were significantly higher in the study group than in the U.S. population when examined in their totality or stratified by "high lead exposure" (>0.2 mg/m^3 lead in air, surveyed in 1975) or "duration of exposure." Imprecise estimation of lead exposure may have contributed to the nonsignificant results.

A study of 664 male workers in a Swedish lead smelter from 1942-1987 examined standardized mortality ratios for cardiovascular disease compared to the county population mortality figures from 1969-1989 (Gerhardsson et al., 1995a). Blood lead measurements were

available from the workers since 1969 (mean 62.1 µg/dL) and dropped steadily from that date to 1985 (mean 33.1 µg/dL). The consecutive blood lead measurements in the subjects allowed construction of a cumulative blood lead index. Standardized mortality ratios were significantly elevated in the group for all cardiovascular diseases (ICD-8 390-458) and for ischemic heart disease (ICD-8 410-414), 1.46 (95% CI: 1.05, 2.02) and 1.72 (95% CI: 1.20, 2.42), respectively. However, there were no indications of a concentration-response relationship when analyses were stratified by cumulative blood lead index, peak blood lead, or other exposure indices.

In a study of 1,261 male newspaper linotype operators working in 1961 and followed until 1984, 38% had died from all causes (Michaels et al., 1991). Compared to the New York City population reference group, there was a marginally significant increased standardized mortality ratio in the printers of 1.35 (95% CI: 0.98, 1.82) for cerebrovascular disease (ICD-8 430-438), which became highly significant in those with 30 or more years exposure (1.68 [95% CI: 1.18, 2.31]; 37 of the total 43 deaths due to cerebrovascular disease). Atherosclerotic heart disease (ICD-8 410-414) mortality in printers was significantly below that expected from the general population, with a standardized mortality ratio of 0.63 (95% CI: 0.59, 0.73).

Two studies were longitudinal in nature, following the same cohort for a period from 12 to 16 years (Lustberg and Silbergeld, 2002; Møller and Kristensen, 1992). They both used large community cohorts (NHANES II and a Danish birth cohort, respectively) and they both used multivariate proportional hazards models with blood lead as the principal predictor. Both studies found a significant increase in risk ratio with increased blood lead.

Another study examined occupationally-exposed subjects and used population reference groups to assess differences in mortality. Steenland et al. (1992) showed no significant increased mortality from ischemic heart disease (ICD-9 410-414), hypertension with heart disease (ICD-9 402 and 404), hypertension with no heart disease (ICD-9 401, 403, and 405), nor cerebrovascular disease (ICD-9 430-438) in the study group than in the U.S. population, even in the high lead exposure group.

6.5.3.4 Other Cardiovascular Effects

Peripheral arterial disease (PAD), flow-limiting atherosclerosis in lower limb muscular arteries, was studied using Phase 1 (1999-2000) of the NHANES IV, the most recent NHANES

dataset (Navas-Acien et al., 2004). PAD was categorized as a ratio of brachial artery (arm) systolic blood pressure to posterior tibial artery (ankle) systolic blood pressure < 0.90. One hundred thirty-nine subjects were classified as having PAD; there were 1,986 subjects without the disease. Blood lead was classified by quartile, with the 1st quartile containing subjects with blood lead <1.4 µg/dL and the 4th quartile containing subjects with blood lead >2.9 µg/dL. Age range was 40 to >70 years. Three sets of covariates were tested in separate models. The first set, common to all models, included age, sex, race, and education. The second set included the first set and added BMI, alcohol intake, hypertension, diabetes, hypercholesterolemia, and glomerular filtration rate. The third set added self-reported smoking status and serum cotinine. Compared to first quartile blood lead, 4th quartile blood lead subjects had significant odds ratios for PAD of 3.78 (95% CI: 1.08, 13.19) and 4.07 (95% CI: 1.21, 13.73) for the first two models. The odds ratio of 2.88 (0.87, 9.47) for the third model was not statistically significant. However, the increasing odds ratio trend from 1st through 4th quartile was significant for all 3 models ($p \geq 0.02$).

6.5.4 Potential Confounding of the Cardiovascular Effects of Lead

6.5.4.1 Confounding by Copollutants

High on the list of other metals that might be associated with cardiovascular disease is cadmium, through its known effects on kidney function. If blood lead and blood cadmium strongly covary in a sample by sharing a common source (e.g., when the study sample is drawn from a population living near a nonferrous smelter emitting both metals), including simultaneous blood lead and cadmium measurements in the same model would likely show a significant reduction in both coefficients when compared to either metal alone. If, however, blood cadmium and lead do not covary in the sample, their coefficients in the model together would be much the same as when tested separately. In a study of PAD (Navas-Acien et al., 2004) discussed in Section 6.5.3.4, investigators not only tested both lead and cadmium in separate models but also tested them simultaneously. The correlation coefficient between natural log lead and natural log cadmium was 0.32 ($p < 0.001$), highly significant, though leaving 90% of the variance between them unexplained. In addition, they tested possible interactions between lead and cadmium, and between the two metals and sex, race-ethnicity, smoking status, renal function, and C-reactive protein. Although none of the interactions were significant, when blood lead and blood cadmium

were in the same model together they both had significant trends of increasing odds ratios with increasing quartile of each metal, but the nonsignificant point estimate of the odds ratio for blood lead comparing 1st and 4th lead quartile tested alone dropped further when tested with cadmium (odds ratio of 2.88 versus 2.52). Cadmium between 1st and 4th quartile, on the other hand, showed a similar drop from cadmium tested alone to cadmium tested with lead (odds ratio of 2.82 versus 2.42), but both point estimates remained significant. Thus, though point estimates of both lead and cadmium were approximately the same whether tested alone or together, the larger variance associated with the lead coefficients rendered them nonsignificant. Part of the difference in variance between the two metals could be explained by noting that the reference group (lowest quartile) for lead contained a little over half the number of subjects (n = 472; 18 cases, 454 noncases) than the reference group for cadmium (n = 856; 27 cases, 829 noncases). The odds ratios for PAD with smoking status dropped from 4.13 (95% CI: 1.87, 9.12) to 3.38 (95% CI: 1.56, 7.35) when lead was added to the model, but both odds ratios remained highly significant and the difference was not statistically tested. The failure to find a significant interaction between the two metals and between smoking status and both metals suggests that none of the odds ratio changes discussed above were significant. The same pattern of results was found when using cotinine blood levels instead of self-reported smoking habit. Adding cadmium alone or cadmium and lead together resulted in nonsignificant odds ratios for both indices of smoking.

The Belgian Cadmibel studies also were ideally situated to test possible interactions between blood lead and cadmium, but the technique of stepwise addition of variables to the multiple regression models of blood pressure did not allow retention of both metal variables together in the same model (Staessen et al., 1996). From the lack of both cadmium and lead in any one model, it can be inferred that, if both variables had been forced into the model together, they both would have had nonsignificant coefficients.

6.5.4.2 Confounding by Smoking Status

Most studies reviewed in this section have controlled for tobacco use, where it often appears related to lower blood pressure. The majority of reviewed studies including smoking as a covariate never present the coefficients of smoking or related covariates. Only the Navas-

Acien et al. (2004) study discussed in the previous section systematically addressed the issues related to possible confounding or effect modification with tobacco use.

6.5.4.3 Confounding by Alcohol Consumption

Possible confounding by alcohol use, generally associated with increased blood pressure, was thoroughly discussed in the 1990 Supplement (Grandjean et al., 1989). Alcohol, especially in Europe, contained substantial lead during much of the 20th century. This can be seen in the MONICA Augsberg, Germany cohort study (Hense et al., 1994). The study group was stratified by sex and then, only in men, by rural-urban location. Within each strata, the blood lead range differed by alcohol use. In women, for example, the 10th and 90th percentile values of blood lead were approximately (as estimated from graphs) 3.5 and 8.5 µg/dL for self-reported abstainers, 4.5 and 10.5 µg/dL in those drinking from 1 to 39 g/day, and 6.0 to 14.0 µg/dL in those drinking 40 plus g/day. Despite the finding that only women in the highest alcohol-use group had a significant lead effect, it cannot be determined if the increase in lead coefficient is significant because the three coefficients associated with use of alcohol strata were not tested for differences among themselves; they were only tested for their significance from the null hypothesis of 0. Another study was based on subjects from the New Risk Factors Survey from the area around Rome, intended to determine confounding effects of a number of social and biochemical variables on the blood lead-blood pressure relationship (Menditto et al., 1994). Alcohol consumption, as well as BMI, heart rate, non-HDL cholesterol, and HDL cholesterol, triglycerides, cigarettes smoked/day, and skinfold thickness were all examined. A doubling of blood lead was associated with an increase of 4.71 mm Hg in systolic and 1.25 mm Hg in diastolic blood pressure. Alcohol as a true confounding variable is likely limited to studies in areas where alcohol contributes significantly to blood lead. In a study of 249 bus drivers in San Francisco, CA, natural log lead coefficients against blood pressure changed less than 10% when alcohol use was included as a covariate (Sharp et al., 1990). Blood lead according to alcohol use was not reported. Another study based on a U.S. population found a significant increase in blood lead of a mixed group of males and females according to alcohol use, ranging from mean blood lead of 7.3 µg/dL in nonusers to 9.2 µg/dL in those reporting more than 2 ounces/day over 3 days (Morris et al., 1990), with no report of significant effects of alcohol on blood pressure.

6.5.4.4 Confounding by Dietary Calcium Intake

The main thrust of the previously reported Morris et al. (1990) study was to examine the effects of dietary calcium on the effect of lead on blood pressure in 78 males and 64 females between 18 and 80 years, many of whom were hypertensive (undisclosed number), though those using medications for hypertension discontinued their use 1 month before testing started. Subjects were excluded if they had "secondary hypertension." The investigators measured serum calcium and assessed dietary calcium intake, among other variables. There were no changes in blood lead or blood pressure noted as a result of dietary calcium supplementation.

Proctor et al. (1996), using the Normative Aging Study, examined possible modification of the effect of natural log blood lead (blood lead range 0.5-35 μg/dL) on blood pressure in 798 men aged 45-93 years by dietary calcium intake assessed by food questionnaire. The study used multiple regression models with a fixed set of covariates, including age and age^2, BMI, adjusted dietary calcium, exercise, smoking, alcohol use, sitting heart rate, and hematocrit. Increased blood lead was significantly associated with diastolic blood pressure and systolic blood pressure. Only systolic blood pressure significantly decreased with increased dietary calcium (0.004 mm Hg decrease for every 1 mg/day increase of dietary calcium). The authors formed dichotomized calcium intake (cut point at 800 mg/day) and blood lead (cut point at 15 μg/dL) variables to test the interaction between blood lead and calcium on blood pressure. They did not find a significant interaction.

A study of a subset of the Cadmibel Study with 827 males and 821 females, age 20 to 88 years, selected from areas known to represent a wide range of cadmium exposure, specifically studied total serum calcium interactions with blood lead on blood pressure (Staessen et al., 1993). Stepwise regression models, selecting from log blood lead, age and age^2, BMI, pulse rate, log serum gamma-glutamyltranspeptidase, serum calcium, log serum creatinine, urinary potassium, smoking, alcohol intake, contraceptive pill use (females only), and a menopause indicator variable (females only), were stratified by sex for systolic and diastolic blood pressure. The stepwise procedure resulted in models each with a different mix of covariates. Increased serum calcium was significantly associated with increased systolic blood pressure in both males and females. Every increase of one log unit of blood lead was associated with nonsignificant changes in blood pressure in women but with a significant decrease in systolic blood pressure in men (systolic log blood lead $\beta = -5.2$). A separate set of models were constructed with an

interaction term between serum calcium and log blood lead (details not shown). In women only, both main effects of lead and calcium and the interaction effect were significant (no coefficients presented). At the 25th percentile of serum calcium (2.31 μmol/L), a doubling of blood lead was associated with a 1.0 mm Hg increase in systolic blood pressure. At the 75th percentile of serum calcium (2.42 μmol/L) a doubling of blood lead was associated with a 1.5 mm Hg increase in systolic blood pressure. Furthermore, serum calcium may itself be confounded with age in women, as women showed a sharp rise in serum calcium in their sixth decade of life, coincident with menopause, whereas the trend for serum calcium in men was steadily downward for each subsequent decade of age. The authors did not test an interaction term including calcium and age or calcium and menopausal status. Thus, the significant interaction effect between calcium and lead on blood pressure may be a result of differences due to menopause.

6.5.4.5 Summary of Potential Confounding of the Lead Effect on Cardiovascular Health

The effects of cadmium exposure, smoking, alcohol use, dietary and serum calcium levels have all been formally tested in a few studies, without significant effects as confounders of the lead effect. Failure to find a significant confounding effect with lead, however, does not argue to maintain these variables uncritically in models of blood pressure. If alcohol contains lead, increased alcohol use will lead to increased blood lead. In this case, both variables in the model will be collinear and this tends to distort estimated coefficients and standard errors of their effect on cardiovascular outcome. Tobacco use may influence lead levels much more in occupational studies than in community exposure studies, especially if smoking in the factory is allowed. Frequent hand to mouth behavior will increase lead exposure and, consequently, raise blood lead concentrations. Serum calcium may statistically modify the lead effect differentially by gender due to menopause in women. Menopause also affects lead turnover. If serum calcium, blood lead, and blood pressure are all statistically related, serum calcium should not be used in blood lead-blood pressure/hypertension studies.

6.5.5 Gene-Lead Interactions

Study authors characterized sodium-potassium adenosine triphosphatase α2 (ATP1A2) polymorphism in 220 workers formerly exposed to a mix of organic and inorganic lead in the U.S., noted above in other references (Glenn et al., 2001). The ATP1A2 (3′) one kilobase probe

produced two homozygous (4.3/4.3 and 10.5/10.5) and one heterozygous (4.3/10.5) genotypes and two homozygous (8.0/8.0 and 3.3/3.3) and one heterozygous (8.0/3.3) genotypes for the 2.5 kilobase ATP1A2 (5′) probe. Of the 209 subjects with data on both polymorphisms, 43.5% were doubly homozygous for 8.0/8.0 and 4.3/4.3, 34.4% were homozygous for 8.0/8.0 and heterozygous for 4.3/10.5, 11.5% were heterozygous for 8.0/3.3 and homozygous for 4.3/4.3, 5.3%. Also, 5.3% were doubly homozygous for 8.0/8.0-10.5/10.5, and 4.8% were doubly heterozygous for the two genotypes. Although only 13 African American workers participated, prevalence of the 10.5 kilobase allele in the ATP1A2 (3′) genotype was statistically higher for them than for other races. Prevalence of hypertension (≥160/96 mm Hg or use of hypertension medication) was significantly higher in those with the 10.5/10.5 genotype than in others. Controlling for age, BMI, lifetime number of alcoholic drinks, the 10.5/10.5 genotype was associated with an odds ratio of 7.7 (95% CI: 1.9, 31.4) for hypertension when compared to the 4.3/4.3 homozygous genotype, but there were no effects of either blood lead, tibia lead, or their interaction with ATP1A2 (3′) genotype. A multiple linear regression model for linear blood lead and systolic blood pressure, controlling for age, use of hypertensive medication, current smoking, quartiles of lifetime alcohol consumption, and season, showed a significant main effect for 10.5/10.5 homozygous contrasted against combined 4.3/4.3 and 4.3/10.5 groups, associated with a 25.5 mm Hg reduction in blood pressure, primarily due to limited blood lead range of the homozygous group (maximum blood lead of the 10.5/10.5 group 9 μg/dL; maximum blood lead of the contrast group = 20 μg/dL). But the interaction between linear blood lead and the 10.5/10.5 condition resulted in a significant increase of the blood lead effect on blood pressure by 5.6 mm Hg for every 1 μg/dL blood lead compared to the blood lead effect in the other genotypes. The authors stated, but did not show analysis or coefficients, that the ATP1A3 (3′) polymorphism also significantly interacted with tibia lead and systolic blood pressure. There were no significant relationships using the ATP1A2 (5′) gene. Thus, the ATP1A2 (3′) polymorphism appears to directly influence both prevalence of hypertension and the effect of lead on blood pressure, though the small group (n = 9 with all measures) with the important 10.5/10.5 homozygous pattern would argue for enlarging this important study.

Another research group focused on polymorphisms of two genes suspected to be involved in lead toxicokinetics, the vitamin D receptor (VDR) and delta-aminolevulinic acid dehydratase (ALAD) (Lee et al., 2001). Polymorphism of both genes is well studied and prevalence appears

associated with race or ethnic background. Nearly 800 Korean workers aged 18-65 years (79.4% males) from lead-using businesses were classified according to ALAD polymorphism (1-1 [homozygous] versus 1-2 [heterozygous]) and VDR polymorphism (bb [predominant homozygous] versus Bb plus BB [infrequent polymorphisms]). The homozygous 1-1 ALAD polymorphism was found in 90.1% of the group and the homozygous bb polymorphism was found in 88.8% of the group. When compared to a smaller group of non-lead-exposed workers, blood lead concentration (mean exposed 32.0 µg/dL [range 4-86] mean nonexposed 5.3 µg/dL [range 2-10] and tibia lead concentration mean exposed 37.2 µg/g [range −7-338]; and mean nonexposed 5.8 µg/dL [range −11-27]) were much higher. The study used stepwise multiple regression models, selecting covariates remaining significant in the models from among a large set of potential control and confounding variables. They also allowed potential confounders to remain in the models if "there were substantive changes in the coefficients of predictor variables" with their addition. Systolic models controlled for age and age^2, sex, BMI, antihypertensive medication use, and cumulative lifetime alcohol use. Depending on the presence or absence of linear blood lead, tibia lead, and DMSA chelatable lead in the models, and the gene-age interactions tested, blood urea was added to the model. Diastolic models controlled for age, sex, BMI, cumulative alcohol consumption, and linear blood lead. Hypertension (systolic >160 mm Hg or diastolic >96 mm Hg) logistic multiple regression models controlled for age, sex, BMI, tibia lead, and current alcohol use. Among the exposed workers bb VDR genotypes had significantly lower DMSA-chelatable blood lead and lower diastolic and systolic blood pressure than the combined Bb and BB genotypes. The only significant interaction reported between predictor variables and gene polymorphism on blood pressure was with the VDR polymorphism bb allele, who had a less pronounced increase in systolic blood pressure with age than subjects with the B allele. There were only marginally significant associations of systolic blood pressure with tibia lead and linear blood lead. There were no significant associations in models of diastolic blood pressure with linear blood lead, DMSA-chelatable blood lead, or tibia lead. Tibia lead was significantly associated with hypertension (odds ratio of 1.05 [95% CI: 1.00, 1.12] for each 10 µg/dL increase in tibia lead). Workers with VDR B allele had significantly higher prevalence of hypertension (odds ratio = 2.1 [95% CI: 1.0, 4.4]) than workers with the bb genotype, but no other lead variable or interaction with VDR status was reported significant. Though VDR status was significantly related to blood

pressure and prevalence of hypertension, there were no significant effects of ALAD polymorphism on blood pressure or hypertension or of VDR interactions with any lead exposure variable.

Lustberg et al. (2004) studied these same Korean lead workers (n = 793) to examine the relationships between the G^{894}–T^{894} polymorphism in the gene regulating endothelial nitric oxide synthase (eNOS) and blood lead effects on blood pressure and hypertension. Nitric oxide metabolism has been suggested both as a mechanism for altered blood pressure and for moderating the effects of lead on blood pressure, though there is experimental support for and against both hypotheses. After classifying subjects as homogenous for the GG type (85%), heterogeneous for both types (TG) (14%), or homogenous for TT (1%), the TG and TT types were combined into a single group (TG/TT). Diastolic and systolic multiple regression models were constructed with a fixed set of covariates, including smoking, alcohol consumption, age, sex, BMI, and education. Logistic regression models used blood pressure criteria of either ≥140 mm Hg diastolic blood pressure, ≥90 mm Hg systolic blood pressure, or self-report of using antihypertensive medications. There was no effect of genotype on diastolic or systolic blood pressure or on hypertension prevalence in multiple regression models, nor any significant interaction of lead exposure indices with gene status.

6.5.6 Summary of the Epidemiologic Evidence for the Cardiovascular Effects of Lead

The combined blood lead studies using blood pressure/hypertension as an outcome continue to support the conclusions of the 1990 Supplement that there is a positive association between blood lead and increased blood pressure. The occasional finding of significant negative associations of blood lead with blood pressure (e.g., the Cadmibel study, one NHANES III study, the postpartum phase of the Los Angeles pregnancy study) have not been adequately explained and require further confirmation and study. The reported meta-analysis succinctly characterizes the blood pressure findings with blood lead: 1.0 mm Hg systolic pressure increase with each doubling of blood lead; 0.6 mm Hg diastolic pressure increase with each doubling of blood lead. Although females often show lower lead coefficients than males, and blacks higher lead coefficients than whites, where these differences have been formally tested, they are usually not statistically significant. The tendencies may well arise in the differential lead exposure in these

strata, lower in women than in men, higher in blacks than in whites. The same sex and race differential is found with blood pressure.

The most promising developments in this field since the 1990 Supplement have been the use of bone lead as a long-term cumulative lead exposure index and the introduction of genetic analysis into the studies as potential lead effect modifiers. With one exception, all studies using bone lead have found a consistently positive and significant effect on blood pressure and/or hypertension. The ability to estimate past exposure in cross-sectional studies is a significant advance. The results of the bone lead studies to date highlight the important role of accumulated lead exposure in the development of cardiovascular problems.

Though the study of genetic polymorphisms is still in its infancy in this field, it too holds great promise in accounting for individual variability in development of cardiovascular disease and individual response to lead exposure.

6.6 REPRODUCTIVE AND DEVELOPMENTAL EFFECTS OF LEAD

6.6.1 Summary of Key Findings of the Reproductive and Developmental Effects of Lead from the 1986 Lead AQCD

Lead has been implicated as a risk factor for reproductive outcomes for over a century Rom, 1976; Oliver, 1911). As early as 1860, increased rates of stillbirths and spontaneous abortions were found in women with occupational exposure to lead (usually in the ceramics industry) and in women with husbands employed in the lead industry, compared to unexposed women (Rom, 1976). Other early investigations found increased rates of physically and mentally "retarded" offspring among these same groups. In 1910, these findings resulted in the first lead-related occupational regulation; the British Committee on Occupational Health recommended that women not be employed in the lead industry (Oliver, 1911). These observations, however, were based on exposure levels far above those considered acceptable today, and current research now focuses on substantially lower exposure levels.

The 1986 Lead AQCD provided evidence that lead, at high exposure levels, exerted significant adverse health effects on male reproductive functions. Several studies observed aberrations in both sperm count and morphology in men occupationally exposed to relatively high levels of lead (blood lead levels of 40-50 μg/dL). However, the effects of lead on female

reproductive function and fetal growth were suggestive but equivocal, perhaps due to the small sample sizes and inadequate controlling for potential confounding factors.

This section provides a critical review of the literature regarding the associations between exposure to environmental lead and reproductive outcomes. First, the evidence for the placental transfer of lead is reviewed; this is key to providing a basis and mechanism for fetal exposure. Second, the association between exposure and each outcome is reviewed. Outcomes of interest are reproductive function (fertility), spontaneous abortion, fetal growth, preterm delivery, and congenital anomalies. Each section below begins with a summary of the literature up to 1986, the year of the last EPA Air Quality Criteria Document. Then, key studies are reviewed and each section ends with a conclusion based on the evidence provided. The conclusion is based on the generally accepted "Causal Criteria" for bodies of epidemiologic literature (Hill, 1965; Susser, 1991).

6.6.2 Placental Transfer of Lead

In 1968, Barltrop (1968) demonstrated that lead crosses the placenta beginning as early as gestational week 12. He found that the rate of transfer subsequently increased to term. Lead accumulations were found in the bones, livers, blood, hearts, kidneys, and brains of stillborn and spontaneously aborted fetuses. These observations were replicated by numerous investigators; for example, Casey and Robinson (1978) found lead accumulations in the livers, kidneys, and brains of stillborn fetuses. Lead accumulations were also found in the livers, brains and kidneys of first trimester abort fetuses (Chaube et al., 1972), suggesting placental transfer earlier than 12 weeks of gestation. Newer findings, published since 1986, are reviewed below (also see Section 6.2.2.5.2).

Placental transfer of lead is confirmed by correlations of maternal blood lead concentrations, umbilical cord blood lead, and placental lead concentrations in a variety of settings. Umbilical cord blood reflects fetal blood. Early studies, prior to 1986, found correlation coefficients between maternal and umbilical cord blood lead ranging from 0.5 to 0.8, all of which were highly statistically significant. More recent studies also find significant correlations between maternal and fetal blood lead. For example, a prospective study in Kosovo, Yugoslavia recruited 1,502 women at mid-pregnancy in two towns — one with high exposure due to the presence of a lead smelter, refinery, and battery plant, and one with relatively low

exposure. The correlation between maternal blood lead (either at delivery or at mid-pregnancy) and cord blood lead ranged from 0.8 to 0.9 (Graziano et al., 1990). Among women with substantially lower levels of exposure (e.g., blood lead 1.9 μg/dL) the correlation between maternal and cord blood lead was 0.79 (Harville et al., 2005).

Chuang et al. (2001) propose that while maternal and cord whole blood lead are highly related, fetal exposure may be even more influenced by maternal plasma lead. Using data from a cohort of 615 women in Mexico City recruited in 1994-1995, these investigators used structural equation modeling to estimate the associations between whole blood lead, bone lead (cortical and trabecular), and the latent variable, plasma lead and cord blood lead. They found the strongest associations between whole blood lead and cord blood lead, even after accounting for plasma lead. The greatest contributors to plasma lead were bone lead and airborne lead. However, with declining exogenous lead exposure, these investigators note that the measurement of plasma and bone lead may become increasingly important in assessing fetal exposure.

These data provide little doubt of fetal exposure to lead via placental transport. Further, it appears that lead crosses the placenta throughout pregnancy, leading to continual exposure of the fetus. Indeed, there is evidence to suggest that maternal blood leads during the later half of gestation increase (Gulson et al., 2004; Hertz-Picciotto et al., 2000; Rothenberg et al., 1994; Sowers et al., 2002). The magnitude of the increase ranges from 14-40%, possibly due to the different starting blood leads in each study (Bellinger, 2005). The increase in blood lead in the later half of pregnancy may result from physiologic changes in maternal homeostasis during pregnancy and, in particular, to mobilization of lead stores from other body organs (Bellinger, 2005). Indirect evidence for such mobilization comes from the increased rate of bone turnover during the later half of gestation, prompted by the increased fetal need for calcium (Moline et al., 2000). Thus, both the epidemiological evidence and the biological plausibility of the associations support the role of maternal-fetal transfer of lead.

Additionally, in populations with greater lead burdens, the fetus may be at even greater increased risk for exposure and possible adverse effects of exposure. Among the variables associated with lead exposure in pregnant (and nonpregnant) women are: smoking and alcohol consumption (Graziano et al., 1990; Rhainds and Levallois, 1997), pica (Rothenberg et al., 1999), use of ethnic remedies and cosmetics (Al-Ashban et al., 2004; Centers for Disease Control and Prevention, 1993), and food preparation in inappropriately lead-glazed pottery

(Azcona-Cruz et al., 2000; Rothenberg et al., 2000). There is some evidence that low calcium intake is also associated with higher blood lead (Gulson et al., 2004; Hernandez-Avila et al., 2003; Hertz-Picciotto et al., 2000). Finally, the location where the mother resides (or resided as a child) may increase blood lead (Graziano et al., 1990). Blood leads are elevated among U.S. immigrants, especially those who migrated from countries where lead is still used as a gasoline additive (Centers for Disease Control and Prevention, 2000); indeed, blood leads are inversely associated with the number of years since migration (Centers for Disease Control and Prevention, 2000; Klitzman et al., 2002; Rothenberg et al., 1999).

In conclusion, the epidemiologic evidence indicates that lead freely crosses the placenta, resulting in continued fetal exposure throughout pregnancy. Indeed, the evidence is strong that exposure increases during the later half of pregnancy. Exposure to the fetus is more pronounced in high-risk populations, especially those who migrated from countries still using lead as a gasoline additive.

6.6.3 Effects of Lead on Reproductive Function

6.6.3.1 Effects on Male Reproductive Function

Male reproductive function is measured using the reproductive history of the male (i.e., number of pregnancies fathered), time to pregnancy and direct measures of semen quality (usually sperm count, motility and morphology). Most studies relating lead exposure to male reproductive function are based on data collected in the occupational setting linked to population birth registries and on studies directly collecting questionnaire exposure and outcome data.

6.6.3.1.1 Sperm Count, Motility and Morphology

Recent publications which purport a decline in sperm concentration, motility, and morphology seek the explanation in the rising use of man-made chemical endocrine disruptors (Auger et al., 1995; Fisch et al., 1997; Farrow, 1994; Gyllenborg et al., 1999; Kavlock et al., 1996; Keiding et al., 1994; Kieding and Skakkebaek, 1996; Lerchl, 1995; Olsen et al., 1995; Sherins, 1995). Several studies from the 1970s and early 1980s suggest aberrations in both sperm count and morphology in men exposed to relatively high levels of lead. In the earliest study, Lancranjan et al. (1975) found decreased sperm counts and an increased prevalence of morphologically abnormal sperm among workers heavily exposed to lead (mean blood lead

74.5 μg/dL) as well as those moderately exposed (mean blood lead 52.8 μg/dL). These findings have been corroborated by results of studies in the U.S. (Cullen et al., 1984) and Italy (Assennato et al., 1986) which describe similar effects in workers with blood leads above 60 μg/dL.

More recently, corroborating data was described in a comprehensive review by Apostoli et al. (1998). In studies of men with blood leads above 40 μg/dL, decreases in sperm count and concentration, motility and morphologic aberrations were found. Chowdhury et al. (1986) found a significant decrease in sperm count and motility and an increase in the number of sperm with abnormal morphology in 10 men with occupational lead exposure; the average blood lead in the exposed group was 42.5 μg/dL compared to 14.8 μg/dL in the unexposed. Similar results were found in a group of 30 lead-exposed factory workers compared to controls (Lerda, 1992). In a large study of male lead smelter workers, Alexander et al. (1996a) found a decreasing trend of sperm concentrations with increasing lead exposure. In this cohort, 152 workers provided blood specimens and 119 also provided semen samples. Geometric mean sperm concentrations were 79.1, 56.5, 62.7, and 44.4 million cells/mL for blood leads of <15, 15-24, 25-39, and ≥40 μg/dL, respectively. Long-term body lead burden was estimated from current blood lead concentrations and historical blood lead monitoring data. Using this measure of long-term lead body burden, a similar trend was found for sperm concentration, total sperm count, and total motile sperm count. No associations were found for sperm morphology or serum concentrations of reproductive hormones. A study of traffic police in Peru, where leaded gasoline is still in use, found decreases in sperm morphology, concentration, motility and viability among men with blood lead ≥40 μg/dL compared to men with blood lead <40 μg/dL.

Using data from an international study of 503 men employed in the lead industry, Bonde et al. (2002) considered the lowest adverse effect level associated with perturbed semen parameters. Median sperm concentration was reduced by 49% in men with blood lead >50 μg/dL; regression analysis indicated a threshold value of 44 μg/dL. These investigators conclude that adverse effects on sperm quality were unlikely at blood leads <45 μg/dL.

In a population of couples undergoing either artificial insemination or in vitro fertilization, Benoff et al. (2003a,b) found higher concentrations of lead in seminal fluid in the male partner among couples who did not conceive, compared to those who did conceive. While not directly measuring the adverse effects of lead on sperm per se, these data suggest a possible mechanism for the transfer of lead from paternal exposure to the fetal environment. Hernandez-Ochoa et al.

(2005) also provide evidence that lead concentrations in seminal fluid may be a better indicator of exposure than blood lead. Mean blood lead in this sample was lower than in most other studies, 9.3 µg/dL. Decreases in sperm concentration, motility, morphology, and viability were correlated with seminal fluid lead or lead in spermatozoa, but not with blood lead.

Overall, the available evidence suggests a small association between exposure to lead, usually in the workplace, and perturbed semen quality. It appears that sperm count and morphology (% normal forms) may be decreased at exposures >45 µg/dL. Future research should focus on studies of men exposed to lower levels of lead, as exposures in the very high range are associated primarily with occupational exposure. These studies should also account for variables known to be associated with semen quality and which may also be associated with exposure, e.g., social class, other environmental exposures such as heat and vibration, and lifestyle variables such as cigarette smoking and alcohol use.

6.6.3.1.2 Time to Pregnancy

Time to pregnancy represents a sensitive measure of fecundity. Time to pregnancy is important because it measures the end effect of perturbed reproductive function. While it is important and necessary to understand the associations between prenatal exposures and endocrine abnormalities and semen characteristics, they represent possible antecedents to the occurrence of pregnancy. Previous reports demonstrate good validity and reliability for reports of time to pregnancy in both males and females and when time to recall has been both long and short (Weinberg et al., 1993, 1994).

One advantage to the use of this parameter, as compared to just an infertility measure, is that it does not require categorization of men into fertile and infertile groups. Among couples that succeed in establishing pregnancy, there is considerable variability in the time between discontinuation of contraception and conception (Weinberg et al., 1994). With the possible exception of cigarette smoking and age, very little is known regarding this intercouple variability. Delays in time to pregnancy may be indicative of a range of reproductive abnormalities of both partners, including impaired gametogenesis, hormonal disruptions, and very early unrecognized pregnancy loss. Time to pregnancy has the menstrual cycle as its natural unit and is thus measured in integer units of menstrual cycles.

Usually, time to the most recent pregnancy is taken as the outcome (Baird et al., 1986). The measure of exposure in these studies usually is the fecundity density ratio, which is similar to an incidence density ratio. Fecundity density ratios can be interpreted as the risk of pregnancy among the exposed during an interval, compared to the risk of pregnancy among the unexposed during the same interval. In such studies, the intervals of interest are menstrual cycles. Fecundity density ratios less than one indicate reduced fecundity (i.e., longer time to pregnancy) among the exposed compared to the unexposed, while those greater than one indicate enhanced fecundity (i.e., shorter time to pregnancy) in the exposed. Usually fecundity density ratios are calculated using discrete time Cox proportional hazards regression models.

Several recent studies evaluate time to pregnancy when the male partner is occupationally exposed to lead. The Asclepios Project, a large European collaborative cross-sectional study, evaluated time to pregnancy in 1,108 men of whom 638 were exposed to lead (Joffe et al., 2003). The reference group consisted of lead workers for whom exposure did not coincide with time of pregnancy. The investigators only included pregnancies which resulted in live births. Fecundity density ratios were 1.12 (95% CI: 0.84, 1.49), 0.96 (95% CI: 0.77, 1.19), 0.88 (95% CI: 0.70, 1.10) and 0.93 (95% CI: 0.76, 1.15) for blood leads <20, 20-29, 30-39, and ≥40 µg/dL, respectively. These results indicate that no association was found between blood lead and delayed time to pregnancy. Similar results were found when duration of exposure or cumulative exposure was used as the exposure metric.

A separate report was published in the Italian group of men included in the Asclepios project (Apostoli et al., 2000). Blood lead at the time closest to conception was used as the measure of exposure. Lead-exposed men (n = 251) who had experienced at least one completed pregnancy were compared to nonexposed men (n = 45). Contrary to what was expected, time to pregnancy was significantly shorter among couples in which the male partner was exposed to lead compared to those in which the male partner was not exposed. In secondary analyses, time to pregnancy was longer among men with the highest blood lead (i.e., ≥40 µg/dL). Limiting the analysis solely to exposed men, time to pregnancy was longer among men with higher blood leads.

Among 502 couples identified by Sallmen (2000) from the Finnish Institute of Occupational Health in which the male partner was exposed to lead, time to pregnancy was reduced among those with blood leads >10 µg/dL compared to those with blood leads

≤10 µg/dL. However, when blood lead was stratified, no concentration-response relationship was found. Fecundity density ratios were 0.92 (95% CI: 0.73, 1.16), 0.89 (95% CI: 0.66, 1.20), 0.58 (95% CI: 0.33, 0.96) and 0.83 (95% CI: 0.50, 1.32) for exposures of 10-20, 21-30, 31-40, and ≥40 µg/dL, respectively. In this study, blood leads close to the time of conception were available on 62% of men, while in 38% it was estimated using blood leads obtained at other points or based on job descriptions.

Among 280 pregnancies in 133 couples in which the male partner was employed in a battery plant, 127 were conceived during exposure while the remainder conceived prior to exposure (Shiau et al., 2004). Time to pregnancy increased with increasing blood lead, especially when blood leads were ≥30 µg/dL. Fecundity density ratios were 0.50 (95% CI: 0.34, 0.74) and 0.38 (95% CI: 0.26, 0.56) for blood leads 30-39 and >39 µg/dL, respectively. In 41 couples, one pregnancy occurred prior to exposure and one during exposure – time to pregnancy during exposure was significantly longer. Of note, this is the only study to estimate decreases in time to pregnancy when blood lead was below 40 µg/dL; time to pregnancy increased by 0.15 months for each 1 µg/dL increase in blood lead between 10 and 40 µg/dL.

6.6.3.1.3 Reproductive History

Population-based birth registries in the Scandanavian countries provide data on medically diagnosed pregnancies. These registries provide a basis for linking occupational data on lead exposure obtained by place and duration of employment or by direct measures of blood lead relative to the timing of marriage or conception. Using a roster of men employed in three battery plants in Denmark, Bonde and Kolstad (1997) matched all births to the 1,349 employees when they were age 20-49 years. A control group of 9,656 men who were not employed in a lead industry was chosen. No associations were found between employment or, among those employed in the lead industry, duration of employment in the lead industry and birth rate.

A similar study in Finland (Sallmen, 2000) examined the association between conception and blood lead among men monitored for occupational exposure at the Finnish Institute of Occupational Health (n = 2,111). Men were categorized as probably exposed and possibly exposed based on their measured blood lead in relation to the time of marriage. A nonexposed group of 681 men with blood lead ≤10 µg/dL was similarly evaluated. Among men in the probable exposure group, the risk of failing to achieve a pregnancy increased with increased

blood lead in a monotonic concentration-response fashion. Compared to the nonexposed, the risk ranged from 1.3 to 1.9 for blood leads 10-20 µg/dL and >50 µg/dL, respectively.

Lin et al. (1996) linked records from the Heavy Metal Registry in New York State to birth certificates from the New York State Office of Vital Statistics for the period 1981 to 1992. Exposure was defined as having at least one blood lead measurement above 25 µg/dL and identified 4,256 men. A reference group of 5,148 men was frequency matched for age and residence. The exposed group had fewer births than expected, and was especially pronounced among men employed in the lead industry for over 5 years.

Among 365 men occupationally exposed to metals, Gennart et al. (1992) identified 74 exposed continuously for more than 1 year and with at least one blood lead measurement >20 µg/dL. Compared to a reference group with no occupational exposure, the probability of at least one live birth was significantly reduced. Fertility decreased with increasing duration of exposure but no concentration-response relationship with blood lead was found (possibly due to the small sample size of exposed men).

A study of men exposed to lead in a French battery plant (Coste et al., 1991) reported no effect on fertility. However, this study did not adequately control for potentially confounding variables, particularly those related to the women. Further, nonexposed workers were defined as those with no blood leads recorded which likely resulted in exposure misclassification.

One potential mechanism to explain the associations between lead exposure and male reproductive outcomes may be through an effect of lead on circulating pituitary and testicular hormones. Several studies have evaluated this hypothesis in groups of workers (Braunstein et al., 1978; Cullen et al., 1984; Erfurth et al., 2001; Ng et al., 1991; Rodamilans et al., 1988). In general these studies find perturbations in concentrations of follicle stimulating hormone, luteinizing hormone, and testosterone. Although many of these studies were limited by small sample sizes, lack of control groups, and admixtures of exposure, taken together, they provide evidence for this possible mechanism.

6.6.3.2 Genotoxicity and Chromosomal Aberrations

The potential genotoxicity and ability to induce chromosomal aberrations speak to the mechanisms by which lead is a potential reproductive toxin. Two possible mechanisms by

which lead may affect reproduction are through affinity with proteins and ability to mimic the actions of calcium (Silbergeld et al., 2000).

Data from occupational studies regarding the effects of lead on chromosomes are contradictory; however, the bulk of evidence suggests that there may indeed be a genotoxic effect. Early studies in occupational groups find associations between lead exposure and increased frequency of sister chromatid exchanges (Grandjean et al., 1983; Huang et al., 1988; Leal-Garza et al., 1986; Maki-Paakkanen et al., 1981). Similar results were found in a group of environmentally-exposed children with blood leads ranging from 30 to 63 µg/dL (Dalpra et al., 1983). Increased frequencies of chromosomal aberrations, particularly chromatid aberrations, were found in battery plant workers and were correlated with increased blood lead (Huang et al., 1988). A more marked increase was found when blood leads were above 50 µg/dL. Other occupational studies find similar associations (Al-Hakkak et al., 1986; Forni et al., 1976, 1980; Nordenson et al., 1978; Schwanitz et al., 1970). Other studies find no evidence of chromosomal aberrations when blood leads ranged from 38 to 120 µg/dL (Bauchinger et al., 1977; Maki-Paakkanen et al., 1981; O'Riordan and Evans, 1974; Schmid et al., 1972; Schwanitz et al., 1975). More recently, two studies in battery plant workers (mean blood lead 40.1 µg/dL) and controls (mean blood lead 9.8 µg/dL) found an increase in high-frequency cells and sister chromatid exchanges among the workers, indicating the cytogenetic toxicity of lead (Duydu et al., 2001, 2005). An increase in sister chromatid exchanges, although not statistically significant, was also found in individuals exposed to lead and/or alcohol and tobacco (Rajah and Ahuja, 1995, 1996). In the Lithuanian populations exposed to either environmental or occupational lead, a higher incidence of sister chromotid exchanges and chromosomal aberrations was found (Lazutka et al., 1999), although these populations were also exposed to other potentially genotoxic substances. Recent data also indicates that lead may inhibit DNA repair responses among lead-exposed workers (Karakaya et al., 2005).

Occupational exposure to lead, particularly when blood leads were high (i.e., over 40 µg/dL) was associated with increased mitotic activity in peripheral lymphocytes and with an increased rate of abnormal mitosis (Forni et al., 1976; Minozzo et al., 2004; Sarto et al., 1978; Schwanitz et al., 1970).

6.6.3.2.1 Issues Concerning Studies of Male Fecundity Related to Lead Exposure

In examining studies of fecundity and fertility, several issues relating to interpretation and bias must be addressed. Infertility usually is defined as 12 months of continuous unprotected intercourse without pregnancy. Fecundity represents both a characteristic of the individuals and a characteristic of a couple, meaning that both partners must be biologically able to procreate. Thus, one possible explanation for observations of reduced fecundity related to occupational lead exposure in the male partner is the exposure he "takes home" via transport of dust on clothing and shoes, ultimately resulting in an effect related to the female partner. Other possible interpretations need to account for measurement error, especially related to the outcomes of reproductive history and time to pregnancy, bias in the selection of subjects for study, and the control for potentially confounding variables.

Both reproductive history and time to pregnancy are subject to errors of recall and rely on the veracity of the subject. Several studies have evaluated recall and veracity of the male partner using the female partner as the "gold standard." In general, these find good reliability between the male and female (Weinberg et al., 1993, 1994). Nevertheless, it is possible, at least for studies using men as the sole informant, that the number of pregnancies a man has fathered is underreported. If reporting is nondifferential with regard to lead exposure, then associations will generally be biased towards the null value; however, since characteristics such as social circumstances, ethnicity, and age may affect both exposure and reporting, it is difficult to evaluate the role of bias.

It was not clear from many of the studies that men with medical conditions which affect fecundity/fertility were excluded. Further, several prescription and over-the-counter medications also affect fecundity as does a history of surgery in the genital area (e.g., varicocele). To the extent that these conditions are related to the absence of employment in lead-industries, then the results may be subject to a type of "healthy worker" effect. Because it is unclear whether many of these studies asked about these conditions, this cannot be ruled out as a possible source of bias.

In retrospective studies it is often useful to use the outcome of the most recent pregnancy in the primary analysis. The reason for this is to reduce any possible recall bias. This type of bias may also be an issue in studies which use occupational registry data, i.e., men may have fathered an additional pregnancy after employment in the industry ceased.

Variables considered potential confounders in studies of fertility and fecundity include sociodemographic characteristics (e.g., age, ethnicity, education, occupation); prenatal and recent lifestyle variables such as cigarette smoking, alcohol use, and medication use; exposures through occupation and hobby, and recent medication use. Also important in these studies is control for factors which may affect the partner's fertility, e.g., cigarette smoking. Many of the studies reviewed did not carefully measure or adjust for confounding variables.

The issues presented above potentially limit the interpretation of results from studies examining the association of lead exposure with male fecundity and fertility. Nevertheless, most studies find small associations between lead exposure at high levels (i.e., ≥45 µg/dL) and slightly reduced male fecundity or fertility.

6.6.3.3 Effects on Female Reproductive Function

Few data directly address the effects of lead exposure on fecundity in the female. A recent retrospective study of time to pregnancy among wives of lead workers provides limited support that lead exposure is associated with increased time to pregnancy. Fecundity density ratios were 0.92 (95% CI: 0.72, 0.16), 0.89 (95% CI: 0.66, 1.20), 0.58 (95% CI: 0.33, 0.96), and 0.83 (95% CI: 0.50, 1.32) for blood leads in the male partners of 10-20, 21-30, 31-38 and ≥39 µg/dL compared to <10 µg/dL, respectively. Note however, that exposure here is measured in the male partners and not the females.

Time to pregnancy was evaluated in 121 women biologically monitored for lead exposure at the Finnish Institute of Occupational Health between 1973 and 1983 (Sallmen et al., 1995). Fecundity did not differ with level of exposure (defined as <10 µg/dL, 10-19 µg/dL and ≥20 µg/dL), but among women with blood leads between 29 and 50 µg/dL, there was a suggestion of reduced fecundity (longer time to pregnancy). However, only a small number of subjects (n = 8) were exposed in this range.

In the limited number of studies, there is little evidence regarding the associations between lead exposure and fertility in the female to draw any conclusions at this time.

6.6.4 Spontaneous Abortion

6.6.4.1 Spontaneous Abortion and Maternal Exposure to Lead

Historical observations suggest increased rates of spontaneous abortion among lead-exposed women, particularly those employed in cottage industries (Rom, 1976). Two early studies in a smelter town in Sweden (Nordstrom et al., 1978a, 1979) suggest elevated rates of spontaneous abortion among female employees at the smelter and among female residents living in close proximity to the smelter. Neither of these studies used biological markers of lead exposure. Moreover, the Swedish smelter study included other exposures such as arsenic, zinc, and cadmium; thus the conclusions for these analyses should be tempered.

In contrast, a prospective study in and around a smelter town in Port Pirie, Australia (McMichael et al., 1986) did not find an association between blood lead concentration and spontaneous abortion. However, it was likely that complete ascertainment of spontaneous abortions was not obtained (Rowland and Wilcox, 1987) since most women were recruited for this study after the first trimester of pregnancy. A retrospective cohort study in two towns in the former Yugoslavia (Murphy et al., 1990) showed no associations between lead exposure and spontaneous abortion in the first reported pregnancy. One of these towns was a smelter town with relatively high lead exposure (at recruitment during mid-pregnancy, the mean blood lead concentration was 17.1 μg/dL, while in the control town the mean blood lead was 5.1 μg/dL). A similar study in Poland (Laudanski et al., 1991) evaluated the association between lead-exposed and nonexposed areas for their reproductive histories. Among women in the exposed areas, 11% reported having at least one prior spontaneous abortion, compared to 19.5% of women in the unexposed areas.

Two studies in Finland (Lindbohm et al., 1991; Taskinen, 1988) used hospital registry data to ascertain women with either spontaneous abortions or livebirths. Either maternal job histories (Taskinen, 1988) or both maternal and paternal job histories were obtained from a registry of occupational blood lead measurements. Neither study found evidence of an association between maternal exposure and spontaneous abortion. In the Lindbohm et al. (1991) study, maternal exposure was extrapolated from the occupation of the father.

In Bulgaria, pregnant women residing in or near lead smelting areas or petrochemical plants were prospectively followed for pregnancy outcomes (Tabacova and Balabaeva, 1993). The investigators compared blood leads in those women with spontaneous abortions and those

without. Blood lead concentrations in cases were significantly higher than in controls (mean blood lead 7.1 µg/dL versus 5.2 µg/dL, respectively). However, this study did not fully describe the selection of women nor the definition for cases.

Women employed by the U.S. Forest Service and exposed to lead-based paint (to mark trees for clearing) were studied using self-reported questionnaires (Driscoll, 1998). Adjustment was made for potential confounders and generalized estimating equations were used to adjust for multiple pregnancies per woman. Significant associations were found for three types of paint containing lead pigment (odds ratios of 4.3 [95% CI: 2.0, 9.3], 2.0 [95% CI: 1.2, 3.3] and 1.8 [95% CI: 1.2, 2.6]). While these findings are intriguing, the response rate was only 59% (with no evaluation of selection bias) and the paint also contained solvents thought to be associated with spontaneous abortions.

Borja-Aburto et al. (1999) examined the association between blood lead concentrations and spontaneous abortions in a nested case-control study using incidence density methods and matching for age, calendar time of study entry, public versus private clinic, and gestational age at study entry. They ascertained 668 women during the first trimester of pregnancy in Mexico City. After contacting women biweekly to update pregnancy status, they found 35 cases (6.4%) of spontaneous abortion among women not lost to follow up. An odds ratio of 1.8 (95% CI: 1.1, 3.1) per 5 µg/dL increase in blood lead was observed after adjustment for spermicide use, active and passive smoking, use of alcohol and coffee, maternal age, education, income, physical activity, hair dye use, use of video display terminals, and medical conditions. Mean blood lead in cases (12.0 µg/dL, range 3.1-29 µg/dL) was slightly higher than in controls (10.1 µg/dL, range 1.3-26 µg/dL). Further, after categorizing blood lead into 5 µg/dL intervals, a concentration-response relationship was evident.

More recently, a small study of 57 female workers in a battery plant in China and 62 controls found that 6 spontaneous abortions occurred in the exposed group, compared to none in the controls (Tang and Zu, 2003). A long-term follow-up of survivors of acute plumbism (Hu, 1991) found increased risk of spontaneous abortions or stillbirths (odds ratio of 1.6 [95% CI: 0.6, 4.0]). Although the study was based on small numbers, the data suggest a persistent association between childhood exposure and outcomes later in life.

A review of eight studies (Borja-Aburto et al., 1999; Driscoll, 1998; Laudanski et al., 1991; Lindbohm et al., 1991; McMichael et al., 1986; Murphy et al., 1990; Tabacova and

Balabaeva, 1993; Taskinen, 1988) evaluating maternal exposure to lead (blood lead >30 μg/dL) and spontaneous abortion concluded that there was little evidence that lead exposure at these relatively high levels was associated with an increased risk in spontaneous abortions (Hertz-Picciotto, 2000). However, Hertz-Picciotto also concluded that methodological difficulties in most of these studies (i.e., small sample sizes, inadequate ascertainment of outcome, and possible residual confounding) limited the confidence in these data. Further, she noted that exposure in many of these studies was either measured in an ecologic fashion or biological measures were available, but they were not ascertained during a biologically meaningful period.

Collectively, there is little evidence to support an association between lead exposure in the female and spontaneous abortion. The only well-designed study which finds an association is that of Borja-Aburto et al. (1999); however, these results need to be confirmed in other populations. Studies of spontaneous abortion need be done carefully to avoid possible bias due to recall, use of pregnancies other than the first, and confounding. Retrospective studies, for example, should take full pregnancy histories, including probing for spontaneous abortions versus induced abortions versus stillbirths. In some cultures, for example, induced abortions are frowned upon and women may report spontaneous abortions instead. Additionally, some women may confuse a stillbirth with spontaneous abortion, especially if she is unable to adequately date her pregnancy using date of last menstrual period. Although the use of the most recent pregnancy may curtail problems of recall, other concerns dictate that the first pregnancy be used in studies of spontaneous abortion because the risk of subsequent spontaneous abortion depends on the history of spontaneous abortion. Finally, while few variables are known confounders of this relationship, the following should be controlled: maternal age, education and other socioeconomic indicators, cigarette smoking, and alcohol use. Several studies of spontaneous abortion did not properly adjust for these potentially confounding variables.

One final concern regards the type of spontaneous abortion. Very early spontaneous abortions, i.e., before a clinical pregnancy is diagnosed, may be missed; assuming, however, that both exposed and unexposed women have the same rates of early spontaneous abortions, this would bias the association towards the null. Indeed, this may be true, as many very early spontaneous abortions may be chromosomally abnormal and probably not attributable to lead exposure.

6.6.4.2 Spontaneous Abortion and Paternal Exposure to Lead

Three studies evaluated paternal exposure to lead and spontaneous abortion. Lindbohm et al. (1991), using national databases to identify pregnancy outcomes among 99,186 births in Finland, found no association between paternal employment in jobs with lead exposure and spontaneous abortion (odds ratio of 0.9 [95% CI: 0.9, 1.0]). In a follow up case-control study (Lindbohm et al., 1991b), they ascertained paternal exposure status during the period of spermatogenesis in 213 cases of spontaneous abortion and 500 controls. Exposure was ascertained using blood lead concentrations measured during spermatogenesis for 6% of men; for the remaining 94%, exposure was estimated using a regression model where the independent variables were blood leads measured either prior to or after the period of spermatogenesis. Blood lead (either measured or estimated) was not associated with spontaneous abortion. When analysis was restricted to men with measured blood lead, blood lead concentrations >30 μg/dL were associated with an increased odds of spontaneous abortion (odds ratio of 3.8 [95% CI: 1.2, 2.0]); however, this result was only based on 12 cases and 6 controls.

The third study (Alexander et al., 1996b) found no association between men employed in a lead smelter and spontaneous abortion. For men with "moderate" exposure jobs the estimated odds ratio was 0.8 (95% CI: 0.5, 1.5) and for those with "high" exposure jobs, the estimated odds ratio was 1.4 (95% CI: 0.7, 2.5). Further when blood lead 1 year prior to the pregnancy was used as the exposure measure, no increased odds of spontaneous abortion was found. These results, however, are based on a low participation rate in eligible workers (37%) and should be interpreted with caution. Overall, the available studies provide little evidence for an association between lead exposure in the male and spontaneous abortions.

6.6.5 Fetal Growth

The results of epidemiologic studies regarding the association between lead exposure and birth weight are inconsistent. Cross-sectional studies (Clark, 1977; Gershanik et al., 1974; Moore et al., 1982; Rajegowda et al., 1972) did not find significant correlations between blood lead and birth weight, nor did a study using placental lead as the exposure variable (Wibberley et al., 1977). A case-control study (Bogden et al., 1978) comparing 25 low birth weight babies (1,500-2,500 grams) to 25 controls (>2,500 grams) matched on maternal age, race and social class found a small, nonsignificant difference in maternal and cord blood leads. Mean maternal

blood lead concentrations were 16.2 ± 4.5 μg/dL and 15.3 ± 5.2 μg/dL and mean cord blood leads were 13.8 ± 4.4 μg/dL and 13.1 ± 4.3 μg/dL in cases and controls, respectively. A further study (Huel et al., 1981) found no differences in maternal and fetal hair lead concentrations between infants born small-for-gestational-age compared to those of normal birth weight.

In 1984, Needleman et al. (1984) reported on a cross-sectional study of 5,183 births of at least 20 weeks gestation in Boston, MA. No associations were found between the proportion of births under 2,500 grams and cord blood lead. Exposure levels in this study were relatively low for the time; cord blood leads ranged from <1 to 35 μg/dL. A reanalysis of these data found no relationship between cord blood lead and birth weight when birth weight was considered as a continuous variable (Bellinger, et al., 1991). However, when birth weight was categorized as low birth weight (<2,500 grams), small for gestational age (<10th percentile for gestational age), or intrauterine growth retarded (>2 standard deviations below the mean for gestational age), relative risks of 1.6 (95% CI: 1.0, 2.6), 1.2 (95% CI: 0.8, 1.6) and 1.9 (95% CI: 1.0, 3.4), respectively, were found for each 10 μg/dL increase in cord blood lead levels. Increased relative risks also were found for cord blood lead levels ≥15 μg/dL, compared to cord blood lead <15 μg/dL; however, only 83 of the 5,183 women had exposures in the high range, resulting in imprecise estimates. These data suggest that lead-related modest reductions in birth weight are perhaps plausible when birth weight is expressed as a function of gestational age.

The prospective study of lead exposure in and around Port Pirie, Australia (McMichael et al., 1986) followed 749 pregnancies of at least 20 weeks duration. Mean maternal blood leads at mid-pregnancy were 10.1 μg/dL and 7.0 μg/dL for women residing in Port Pirie and the surrounding communities, respectively. After excluding 9 sets of twins and 10 cases for which the maternal last menstrual period could not be ascertained, no relationship was found between either cord blood lead or maternal blood lead measured at mid-pregnancy or at delivery and birth weight in a multivariate regression model controlling for known determinants of birth weight.

A prospective study in two towns in Kosovo, Yugoslavia evaluated relationships between birth weight (adjusted for gestational age using last menstrual period) and (a) maternal blood lead at mid-pregnancy and delivery and (b) cord blood lead (Factor-Litvak et al., 1991). The towns were vastly different in exposure patterns, as one was the site of a lead smelter, refinery and battery plant (n = 401, mean mid-pregnancy blood lead 19.0 μg/dL) and one was relatively unexposed (n = 506, mean mid-pregnancy blood lead 5.6 μg/dL). No associations were found

between any of the biomarkers of lead and birth weight in either crude analyses or analyses adjusted for potentially confounding variables.

While the aforementioned studies generally found no association between environmental lead exposure and birth weight, three other studies have shown large reductions in birth weight related to lead exposure. These studies, however, have questionable study designs. Nordstrom et al. (1978b, 1979) in a series of ecologic analyses known as the Swedish Smelter Study, found significant reductions in birth weight between the offspring of women either working at or living in close proximity to the smelter. The 125 gram deficit in birth weight among the offspring of women living closest to the smelter was confined to those with parity three or more, an observation which does not appear to be biologically plausible. Moreover, the ecological nature of the study did not allow for individual measurements of blood lead or for control of potentially confounding variables. Hence, while suggestive, these data do not provide strong evidence for a causal association between lead exposure and birth weight.

In a cross-sectional study of 100 "normal" singleton births, a negative correlation was found between placental lead concentration and birth weight (Ward et al., 1987). Mean placental lead concentration in 21 infants weighing less than 3,000 grams was 2.35 ± 0.9 µg/g compared to 1.12 ± 0.4 µg/g in 10 infants weighting more than 4,000 grams. This study has several limitations. First, no statistical adjustment was made for multiple comparisons (many exposures were studied). Second, potentially confounding variables were not controlled. Third, only 31 of the 100 infants, representing the extremes of the birth weight distribution, were studied. Hence, this study also does not provide strong evidence for an association.

In Cincinnati, OH, the association between lead exposure and birth weight was examined in offspring of a cohort of young (mean maternal age = 22.7 years), inner city women, 85% African American, 86% on public assistance, with a mean IQ of 75 (Dietrich et al., 1987a). The mean gestational period of the neonates, as determined by physical examination, was 39.5 weeks. A decrement in birth weight of 172 grams was associated with an increase in blood lead from 10 to 30 µg/dL. Lead exposure in this group was relatively low with a mean blood lead of 8.0 ± 3.7 µg/dL. In a sample of women from this cohort, the interaction between blood lead and maternal age was significantly associated with birth weight; the effect varied from a decrease of 64 grams for 18 year old mothers to 660 grams for 30 year old mothers, as blood lead rose from 10 to 30 µg/dL (Bornschein et al., 1989). Although the Cincinnati study is highly

suggestive of an effect (especially an effect which varies by maternal age) three factors should be considered in the interpretation of their findings. First, length of gestation was estimated by examining the neurological and physical maturity of the neonate (Ballard et al., 1979); other investigators find assessment of gestational age using this scale overestimates gestational age in preterm infants (Constantine et al., 1987; Kramer et al., 1988; Shukla et al., 1987; Spinnato et al., 1984). Second, it is possible that the association between lead and birth weight differs by maternal characteristics such as race, ethnicity, and SES; however, no study has provided a population sufficiently heterogeneous to examine this possible source of difference. Finally, it is possible that confounding by unmeasured maternal lifestyle characteristics may account for the reported association.

A hospital-based study of cord blood lead and pregnancy outcomes in Quebec, Canada, between June 1993 and January 1995 found a slight increase in cord blood lead levels among infants with birth weight <2,500 grams (Rhainds et al., 1999). For those infants with birth weight <2,500 grams, the geometric mean blood lead was 1.8 μg/dL (95% CI: 1.6, 2.9) compared to 1.6 μg/dL (95% CI: 1.5, 1.7), 1.6 μg/dL (95% CI: 1.5, 1.7), and 1.5 μg/dL (95% CI: 1.5, 1.6) among those with birth weights 2,500-2,990, 3,000-3,499, and ≥3,500 grams, respectively. Although suggestive, the study did not control for potentially confounding variables. The investigators also measured cord blood levels of mercury and organochlorine compounds, and observed that mean levels of these toxicants were higher as well in infants who weighed <2,500 g.

More recently Irgens et al. (1998) using data from the Norwegian birth registry found that women occupationally exposed to lead (none/low compared to moderate/high) were more likely to deliver a low birth weight infant (odds ratio of 1.3 [95% CI: 1.1, 1.6]). No association was found for paternal occupational lead exposure. Parental occupational exposure to lead was not associated with low birth weight in the Baltimore-Washington Infant Study database (Min et al., 1996), although subgroup analysis suggested that high paternal exposure may be associated with small-for-gestational-age infants (odds ratio of 2.9 [95% CI: 0.9, 9.2]). Similar findings were reported by Lin et al. (1998) who compared offspring of lead-exposed workers with those of bus drivers. No associations were reported between lead exposure and low birth weight except among the group of men with blood lead levels >25 μg/dL for over 5 years (relative risk of 3.4 [95% CI: 1.4, 8.4]).

Using bone lead as the metric of exposure, Gonzalez-Cossio et al. (1997) found associations with tibia bone lead (but not with patella bone lead or umbilical cord blood lead) and reduced birth weight. Bone lead was measured one month after delivery. Infants with tibia bone lead in the highest quartile (≥15.15 μg lead / g bone mineral) were, on average, 156 g lighter than those in the lowest quartile (≤4.50 μg lead / g bone mineral). Further analyses of these data (Hernandez-Avila et al., 2002) found an association between infants in the highest quintile of tibia bone lead and shorter birth length (odds ratio of 1.8 [95% CI: 1.1, 3.2]).

Two studies have considered the relationship between lead exposure and head circumference (Hernandez-Avila et al., 2002). Among 233 women in Mexico City, high maternal patella bone lead was associated with increased risk of a low head circumference score at delivery (1.02 per μg lead / g bone mineral [95% CI: 1.01, 1.04]). Similar findings were reported by Rothenberg et al. (1999) who found a reduction in six-month head circumference of 1.9 cm (95% CI: 0.9, 3.0) as maternal blood lead rose from 1 to 35 μg/dL. This study, however was plagued by multiple comparisons as head circumference was measured nine times and prenatal blood lead six times – only one statistically significant result was found.

Potential confounders need to be adjusted for to properly assess the relationship between lead exposure and fetal growth. Factors consistently associated with fetal growth include gender, ethnic origin, maternal body build (i.e., pre-pregnancy weight, height), parity, SES, gestational weight gain and nutritional intake during pregnancy, maternal illness, and cigarette smoking (Kramer, 1987). Factors with less established associations include alcohol consumption (Kline et al., 1987; Kramer, 1987) and street drug use (Kline et al., 1987; Kramer, 1987; Zuckerman et al., 1989). To the extent that these factors are associated with blood lead as well as with fetal growth, they must be accounted for in the analysis.

Studies to date are inconsistent regarding the association between lead exposure and birth weight. Several large prospective studies find no association (Factor-Litvak et al., 1991; McMichael et al., 1990), while at least one (Bornschein et al., 1989) did find an association in specific subgroups of women. However, there is limited evidence (Bellinger et al., 1991) for an association between lead exposure and low birth weight (i.e., <2,500 g), small for gestational age (i.e., <10th percentile for gestational age), and intrauterine growth retardation (i.e., >2 standard deviations below the mean for gestational age). These prospective studies were all well-conducted, adequately measured exposure and outcome, and controlled for potential confounding

variables. They did, however, take place in very different populations, suggesting that the association between lead and fetal growth may depend on the population being studied. The Yugoslavia study (Factor-Litvak et al., 1991) took place in two towns in Kosovo, Yugoslavia, which were divergent on exposure and somewhat comparable on other variables. The Port Pirie study took place in a middle class area of Australia (McMichael et al., 1986). The Boston study (Bellinger et al., 1991) took place in a range of social strata in Boston; the exposure in the highest social group was attributable to renovation of older housing stock. Finally, in the Cincinnati study (Bornschein et al., 1989), the study sample was comprised of lower social class African Americans; the mean IQ of the mothers was 75. It is possible that in this latter study, there was some unmeasured variable which accounts for the observed interaction. Thus, the evidence suggests at most a small effect of lead exposure on birth weight and possibly a small association between lead exposure and several dichotomized measures of fetal growth.

6.6.6 Preterm Delivery

Early evidence regarding an association between environmental lead exposure and preterm delivery was inconsistent. In 1976, Fahim et al. found a preterm delivery rate of 13% in 254 pregnant women living near a lead mining community in Missouri, compared to 3% in 249 women living in a control location. These investigators also found higher concentrations of lead in amniotic membrane, but not higher placental or cord lead in preterm compared to term deliveries, regardless of the women's residential locale. This observation prompted other studies of lead and preterm delivery.

Of the cross-sectional studies, the three which show no association employed cord blood lead as the exposure measure and restricted gestational age (Angell and Lavery, 1982; Bellinger et al., 1991; Needleman et al., 1984; Rajegowda et al., 1972). In contrast, three other studies used different exposure markers (placental lead, maternal and cord blood lead, and maternal and fetal hair lead) and found statistically significant associations (Huel et al., 1981; Moore et al., 1982; Ward et al., 1987). Other studies evaluated pregnancy outcomes in relation to maternal delivery blood lead (McMichael et al., 1986; Rahman and Hakeem, 2003).

Of the prospective studies, the Cincinnati study (Bornschein et al., 1989) found no association between both maternal blood lead at mid-pregnancy or maternal blood lead during the neonatal period (10 days post delivery) and preterm delivery. However, gestational age was

estimated by examining the neurological and physical maturity of the neonates (which tends to overestimate gestational age) and not actual dates. In Port Pirie, Australia (McMichael et al., 1986), a concentration-response relationship between maternal delivery blood lead and preterm delivery was reported. Odds ratios ranged from 2.1 to 4.4 in women with blood leads of 7.7-10.6 μg/dL and >13.5 μg/dL, respectively, compared to those with blood lead <7.7 μg/dL. Savitz et al. (1990) used data from the National Natality Survey and found an odds ratio of 2.3 (95% CI: 0.7, 7.0) between maternal occupational exposure to lead and preterm delivery; however, the estimated odds ratio was based on only 7 cases. In the Yugoslavia study (Factor-Litvak et al., 1991) no associations were found between cord blood lead or blood lead measured at mid pregnancy or delivery and either preterm delivery (defined as delivery <37 completed weeks) or gestational age. A registry study in Norway (Irgens et al., 1998) which linked births between 1970 and 1993 to census-based occupation records found a slightly increased odds of preterm delivery among moderate/high lead-exposed women, compared to those with no or low exposure (odds ratio of 1.13 [95% CI: 0.98, 1.29]). Paternal exposure was not found to increase the risk of preterm birth.

An ecologic study in Canada (Philion et al., 1997) examined 30 years of birth records, corresponding to 9,329 births in a smelter city and a control city. Outcome variables were intrauterine growth retardation defined as small for gestational age. The odds ratio for intrauterine growth retardation in the smelter city compared to the control city was 0.83. Further analysis, stratifying time into 5-year intervals also revealed no associations.

A case control study in Mexico City (Torres-Sanchez et al., 1999) evaluated 161 preterm births and 459 full term births. Cord blood lead was significantly higher in the preterm group (9.8 ± 2.0 μg/dL) compared to the full term group (8.4 ± 2.2 μg/dL) only among primiparous women.

Using data from the Baltimore-Washington Infant Study database, Min et al. (1996) found a small association between paternal occupational exposure in the high range and preterm delivery with appropriate weight for gestational age (odds ratio of 2.1 [95% CI: 0.7, 6.5]) and preterm delivery with small for gestational age (odds ratio of 2.4 [95% CI: 1.9, 3.1]). Similar findings were reported by Lin et al. (1998). Comparing the offspring of lead exposed workers with those of bus drivers, they found an elevated relative risk for preterm delivery (3.0 [95% CI: 1.6, 6.8]) only among men with blood leads >25 μg/dL for over 5 years.

In contrast to fetal growth, few factors are consistently related to preterm delivery; thus in both developed and developing countries the majority of preterm deliveries remain unexplained (Kramer 1987; van den Berg and Oechsli, 1984). Factors which are inconsistently associated with preterm delivery include maternal age, SES, pre-pregnant weight, prior history of preterm delivery or spontaneous abortion, and cigarette smoking (Kline et al., 1987; Kramer, 1987). Thus, these factors must be evaluated as potentially confounding factors in studies of lead exposure and preterm delivery.

For preterm delivery, or reduced length of gestation, the evidence for an association with lead exposure is contradictory. Several of the prospective studies find no evidence of an association (Bornschein et al., 1989; Factor-Litvak et al., 1991) while one finds a concentration-response relationship (McMichael et al., 1986). Further, two well-done registry studies (Irgens et al., 1998; Savitz et al., 1990) find some evidence of an association, albeit the number of exposed cases was small. It seems unlikely that the association between lead exposure and preterm delivery is large, but, more research is clearly necessary.

6.6.7 Congenital Abnormalities

Needleman et al. (1984) found an association between cord blood lead and minor congenital anomalies among 4,354 infants born in a single hospital in Boston, MA. All data were obtained from hospital records, not from direct examination of the infants. The most common anomalies were hemangiomas, lymphangiomas, minor skin problems (tags and papillae), and undescended testicles. Blood lead levels were not found to be associated with individual anomalies.

More recently, a number of studies have considered parental lead related to occupational exposure and risk of congenital anomalies in the offspring. In Finland, Sallmen et al. (1992) evaluated the associations between congenital malformations and paternal exposure during the time of spermatogenesis. The overall estimated unadjusted odds ratio for men with blood lead levels >20 μg/dL was 2.4 (95% CI: 0.9, 6.5). Due to small sample sizes, the investigators could only adjust for one potentially confounding factor at a time; this resulted in odds ratios ranging from 1.9 to 3.2. Of note is the lack of consistency of malformations among the five men with the highest blood lead. The malformation observed included congenital heart disease, oral cleft, club foot, polydactyly, and anomalies of the adrenal gland. The breadth of these anomalies suggests

either that lead affects physical development throughout gestation or that this association represents a chance finding. Among 2,021 pregnancies, Alexander et al. (1996b) found slightly elevated odds ratios for congenital defects among men in the lead smelting industry with moderate exposure (odds ratio of 1.9 [95% CI: 0.6, 6.3]) and high exposure (odds ratio of 2.7 [95% CI: 0.7, 9.6]). These estimates are based on 30 birth defects and 12 stillbirths. No analyses were presented which considered individual birth defects. In Norway, neither maternal (odds ratio of 1.25 [95% CI: 0.8, 1.9]) nor paternal (odds ratio of 0.94 [95% CI: 0.8, 1.1]) occupational lead exposure was associated with serious birth defects (Irgens et al., 1998). Similar results were reported by Kristensen et al. (1993) between paternal lead exposure and birth defects, with the exception of a fourfold increase in the risk of cleft lip among male offspring.

The risk of parental lead exposure and neural tube defects was evaluated in a case-control study of 88,449 births (363 neural tube defects) over a 25-year period in Fylde, England (Bound et al., 1997). Women living in areas in which the water lead concentration was >10 μg/L were more likely to deliver a child with a neural tube defect. The association was consistent for anencephaly (n = 169) and spina bifida/cranium bifidum (n = 195), even after adjusting for social class. These authors posit that the association could be a direct effect of lead on neural tube closure or an indirect effect, the latter meaning a reduction in uptake of zinc (due to lead exposure) leading to a reduction in folate uptake. Irgens et al. (1998) partially confirmed these effects on neural tube defects in mothers occupationally-exposed to lead (relative risk of 2.87 [95% CI: 1.05, 6.38]), but not for paternal lead exposure.

The association between total anomalous pulmonary venous return and parental lead exposure during pregnancy (self reported, obtained from industrial hygiene measures, or from a job exposure matrix) was examined in the Baltimore-Washington Infant Study (Jackson et al., 2004). In this case-control study, maternal periconceptional (i.e., 3 months prior to conception through the first trimester) exposure to lead resulted in an estimated odds ratio of 1.57 (95% CI: 0.64, 3.47). For lead-exposed men, the estimated odds ratio was 1.83 (95% CI: 1.00, 3.42). Findings from this study support a possible association between paternal lead exposure and total anomalous pulmonary venous return**.**

Taken together, the evidence suggests few associations between periconceptional or prenatal exposure to lead and congenital anomalies. There is a suggestion of small associations

with high levels of exposure, but many of those studies relied on occupational histories rather than on actual measures of blood lead levels.

6.6.8 Summary of the Epidemiologic Evidence for the Reproductive and Developmental Effects of Lead

There is little doubt that maternal to fetal transmission of lead results from placental transport. This transport occurs throughout pregnancy and may increase in the later stages. Further, there may be populations with increased fetal susceptibility, including populations with high rates of smoking and alcohol use, those using ethnic remedies and cosmetics, and those who use lead glazed pottery. Low levels of calcium intake may also increase fetal exposure.

The available evidence suggests small associations between exposure to lead and male reproductive outcomes. These include perturbed semen quality and increased time to pregnancy. These associations appear at blood lead levels greater the 45 μg/dL, as most studies only considered exposure in the occupational setting. More research is needed regarding possible male reproductive effects at exposure levels in the lower (and currently more relevant) range. There are no adequate data to evaluate associations between lead exposure and female fertility.

With one exception, there is no evidence to suggest an association between either maternal or paternal lead exposure and increased risk of spontaneous abortions. One study in Mexico where the mean maternal blood leads were in the moderate range (i.e. 10-12 μg/dL) suggests an association.

To date, the evidence suggests at most a small association between lead exposure and birth weight and possibly small associations between lead exposure and several dichotomized measures of fetal growth. The reviewed studies occurred in very different populations, and the small associations may reflect some unmeasured or unknown confounding variable. It is unlikely that further epidemiologic research will fully resolve this question. However, several factors, such as maternal SES, maternal education, smoking prevention and reduced use of alcohol, related to lead exposure are associated with increases in birth weight (and decreases in blood lead) and are candidates for intervention.

Similarly, the evidence suggests at most a small association between lead exposure and preterm delivery or reduced length of gestation. The available data also suggest limited associations between either periconceptional or prenatal lead exposure and congenital anomalies.

There is a suggestion of small associations with high levels of exposure, but many of those studies relied on occupational histories rather than on actual measures of blood lead.

Overall, since the 1986 Lead AQCD, a substantial body of work has evaluated the associations between lead exposure and reproductive outcomes. It is now clear that lead clearly crosses the placenta during all trimesters and maternal exposure results in fetal exposure. For many other outcomes, the observed associations are relatively small, especially at the levels of exposure that are currently of interest. Nevertheless, there may be populations that are highly susceptible to lead-related reproductive effects, especially if they have additional risk factors for these outcomes.

6.7 GENOTOXIC AND CARCINOGENIC EFFECTS OF LEAD

6.7.1 Summary of Key Findings from the 1986 Lead AQCD

The 1986 EPA Lead AQCD reviewed five epidemiologic studies of occupationally exposed workers (Cooper and Gaffey, 1975; Davies, 1984; Selevan et al., 1985; Sheffet et al., 1982; McMichael and Johnson, 1982). These workers were exposed to inorganic lead compounds such as lead oxides and lead sulfides. The EPA noted that Cooper and Gaffey reported a significant increase in lung and gastrointestinal cancer among battery and smelter workers in the U.S. (standardized mortality ratios of 1.50 and 1.48 respectively among smelter workers, and 1.32 and 1.23 among battery workers). The EPA further noted that much of this exposure was by inhalation and ingestion of lead oxides, which are relatively insoluble, adding some plausibility to the occurrence of cancer at these two sites. Sheffet et al. (1982) provided some corroborating evidence for gastrointestinal cancer by finding a nonsignificant excess of stomach cancer among U.S. lead chromate pigment workers. However, Davies (1984) did not find any cancer excess among lead chromate pigment workers in the U.K. The EPA noted that Selevan (1984) found a significant excess of kidney cancer among U.S. lead smelter workers based on 6 cases. This finding was judged striking because it mimicked the findings of kidney cancer in animals. The EPA judged that the McMichael and Johnson (1982) study of lead poisoned workers was not particularly informative because the non-poisoned workers may have had substantial lead exposure and no details were given on how lead poisoning was determined.

In summary the EPA felt the evidence was insufficient, stating that "little can now be reliably concluded from available epidemiologic studies."

The studies by Cooper and Gaffey (1975) and Selevan (1984), which are both important because they are large occupational cohorts with documented high exposure, have been updated and are further reviewed below. A cohort study of U.K. battery workers (Malcolm and Barnett, 1982) is also reviewed below.

EPA in 1986 also presented data on human cytogenetic studies, reproducing data from an earlier 1980 International Agency for Research on Cancer (IARC) monograph for metals and metallic compounds (IARC, 1980). For lead, 10 chromosomal aberration studies were judged to be "positive" and 6 such studies were judged to be "negative." On the whole the EPA considered that "under certain conditions lead compounds are capable of inducing chromosomal aberrations in vivo and in tissue cultures." The EPA also reviewed more limited data from two human studies of sister chromatid exchange (Dalpra et al., 1983; Grandjean et al., 1983), one of which was positive and one negative.

6.7.2 Summary of Key Findings by the International Agency for Research on Cancer and the National Toxicology Program

IARC reviewed inorganic and organic lead compounds in its monograph number 87 in February of 2004 (IARC, 2005), and IARC concluded that inorganic lead compounds were probable human carcinogens (Group IIA). This classification is one step down from a classification as a "definite" human carcinogen (Group I). The IARC classification of inorganic lead compounds as probable human carcinogens was based on limited evidence in humans and sufficient evidence in animals. Also, IARC noted that there was insufficient information regarding organic lead compounds (e.g., tetraethyl lead) to make any judgment.

Regarding the human studies, IARC based its evaluation largely on six occupational cohort studies of highly-exposed workers, which were felt to be particularly informative (battery workers in U.S. and U.K., smelter workers in Italy, Sweden, and the U.S.). The IARC assessment focused on four cancer sites, lung, stomach, kidney, and brain. IARC noted that lung showed a significant elevation in one study (Lundstrom et al., 1997) and nonsignificant elevations in a number of others. However, the significant elevation of lung cancer in Lundstrom et al. appeared to be inextricably associated with arsenic in addition to lead exposure

(Englyst et al., 2001). IARC concluded that the strongest epidemiologic evidence for lead carcinogenicity was for stomach cancer, noting that four cohort studies showed a consistent 30-50% excess of stomach cancer vs. external referent populations. IARC noted that confounding by ethnicity, diet, *Helicobacter pylori* infections, or SES may have played a role in the stomach cancer excesses. Finally, IARC noted that while one cohort study showed a 2-fold excess of renal cancer (Steenland et al., 1992), the other studies showed no excess. Similarly, there were no consistent excesses of brain cancer, although one study did find a significant positive dose-response between glioma and blood leads, based on small numbers (Anttila et al., 1996).

The National Toxicology Program (NTP) in 2003 evaluated the carcinogenicity of lead and lead compounds. A summary of its evaluation can be found in NTPs Report on Carcinogens (www.ntp.niehs.nih.gov/ntp/roc/eleventh/profiles/s101lead.pdf), and the detailed evaluation is also available (NTP, 2004). NTP, like IARC, concluded that "lead and lead compounds are reasonably anticipated to be human carcinogens based on limited evidence from studies in humans and sufficient evidence from studies in experimental animals." The NTP considered that "the strongest epidemiologic evidence was for lung and stomach cancer, which are consistently but weakly associated with occupational and industries entailing lead exposure and with indices of individual lead exposure, including job history and biological monitoring of occupationally exposed and general populations. However, most studies of lead exposure and cancer reviewed had limitations, including poor exposure assessment and failure to control for confounders (other factors that could increase the risk of cancer, including lifestyle factors and concurrent occupational exposure to other carcinogens), and did not demonstrate relationships between the amount of exposure (concentration or duration, for example) and the magnitude of cancer risk." NTP, like IARC, also relied heavily on occupational cohort studies in its evaluation of the epidemiologic evidence. NTP (2003) noted that "the mechanisms by which lead causes cancer are not understood. Lead compounds do not appear to cause genetic damage directly, but may do so through several indirect mechanisms, including inhibition of DNA synthesis and repair, oxidative damage, and interaction with DNA-binding proteins and tumor-suppressor proteins."

Both the IARC and NTP evaluations of human evidence relied primarily on occupational studies of highly exposed workers, in which limited evidence of stomach and to some extent lung carcinogenicity was found. There are seven such studies with relatively large populations

(Anttila et al., 1995; Carta et al., 2005; Fanning, 1988; Gerhardsson et al., 1995a; Lundstrom et al., 1997; Steenland et al., 1992; Wong and Harris, 2000). A further study (Ades and Kazantzis, 1988) also addresses lead exposure in a large occupational cohort, although it is compromised by the strong correlation between arsenic and lead exposure in the cohort. It should be noted that the blood lead levels among these workers were generally three to five times higher than the blood lead levels in the two studies of the general U.S. population (Jemal et al., 2002; Lustberg and Silbergeld, 2002; both based on NHANES II) with environmental exposures. For example, mean blood levels in two studies of U.S. lead smelter workers averaged 56 µg/dL in Steenland et al. (1990) in 1976 and 80 µg/dL in Cooper et al. (1985) during the period 1947-1972. In contrast, blood levels in the U.S. population enrolled in NHANES II in late 1976-1980 averaged 14 µg/dL. General population blood lead levels have decreased markedly since the 1970s in many industrial countries with the banning of leaded gasoline. For example, in the U.S. in the early 1990s general population levels averaged 3 µg/dL according to NHANES III (www.atsdr.cdc.gov/toxprofiles/, see lead toxicological profile, page 409). Regarding the occupational studies, while exposure is well documented, exposure-response data are generally not available, making impossible any quantitative inference about likely cancer effects in low exposure groups based on these studies. The high exposure occupational cohorts are the most informative for deciding whether lead is likely to cause cancer, simply because high doses are more likely to show detectable effects than low doses, if effects exist. If lead does cause cancer, and assuming there is no threshold below which exposure does not cause cancer (which is generally true for human carcinogens), current low level exposures to the general public may result in some level of cancers related to lead exposure due to the potential exposure of a large number of people.

6.7.3 Meta-Analyses of Lead and Cancer

There have been two published meta-analyses of the carcinogenicity of lead and lead compounds. The major findings of these studies are summarized in Table 6-7.1. Steenland and Boffeta (2000) relied on eight occupational cohort studies of highly-exposed workers (seven cohort studies, one nested case-control), all of which had documentation of exposure levels. Meta-analyses were conducted for lung, stomach, kidney, and brain cancer. The combined lung cancer relative risk relative risk was 1.30 (95% CI: 1.15, 1.46), based on 675 lung cancer deaths.

Table 6-7.1. Results of Meta-Analyses Addressing the Association Between Lead Exposure and Cancer

	Risk Estimate (95% CI) for indicated outcome [Number of studies utilized in estimate]		
Meta-Analysis	**Lung Cancer**	**Stomach Cancer**	**Renal Cancer**
Fu and Boffetta (1995)	1.24 (1.16, 1.33) [n = 15]	1.33 (1.18, 1.49) [n = 10]	1.19 (0.96, 1.48) [n = 5]
Fu and Boffetta (1995)	1.42 (1.05, 1.92) [battery/smelter only]	1.50 (1.23, 1.83) [battery/smelter only]	1.26 (0.70, 2.26) [battery/smelter only]
Steenland and Boffetta (2000)	1.30 (1.15, 1.46) [n = 8 – cohort only]	1.34 (1.14, 1.57) [n = 8 – cohort only]	1.01 (0.72, 1.42) [n = 7 – cohort only]

However, the authors noted that the lung cancer findings were not consistent across studies, and were influenced highly by one study (Lundstrom et al., 1997) in which confounding by arsenic was likely. Exclusion of this study dropped the combined lung cancer relative risk to 1.14 (95% CI: 1.04, 1.73). The strongest positive evidence was for stomach cancer (relative risk 1.34 [95% CI: 1.14, 1.57], 181 observed deaths). There was little positive evidence for renal cancer (relative risk 1.01 [95% CI: 0.72, 1.42], 40 deaths), or brain cancer (relative risk 1.06 [95% CI: 0.81, 1.40]). All meta-analyses used fixed effects models, given that no evidence of heterogeneity was found across studies (there was significant heterogeneity for lung cancer when the Lundstrom et al. study was included, but not when it was excluded).

Fu and Boffetta (1995) conducted an earlier meta-analysis in which they reviewed 16 cohort and 7 case-control studies. Different numbers of studies were used for meta-analyses of different outcomes, dependent on whether that outcome was reported separately, among other factors. These authors focused their analysis on the occupational studies. Twelve of these studies were used in a meta-analysis of lung cancer, resulting in a combined relative risk of 1.29 (95% CI: 1.10, 1.50) (random effects model). There was significant heterogeneity of lung cancer results across studies. Meta-analyses using fixed effects (no significant heterogeneity between studies) resulted in relative risks of 1.33 (95% CI: 1.18, 1.49) for stomach cancer (10 studies), of 1.19 (95% CI: 0.96, 1.48) for kidney cancer (5 studies), and 1.41 (95% CI: 1.16, 1.71) for bladder cancer (5 studies). No meta-analysis was conducted for brain cancer. Separate analyses for stomach, lung, and kidney cancer were also conducted for those studies with the

highest occupational exposure to lead (3 to 5 studies of battery and smelter workers), which resulted in slightly higher relative risks. The authors concluded that "the findings from the workers with heavy exposure to lead provided some evidence to support the hypothesis of an association between stomach and lung cancer and exposure to lead. The main limitation of the present analysis is that the excess risks do not take account of potential confounders, because little information was available for other occupational exposures, smoking, and dietary habits. The excess risk of stomach cancer may also be explained, at least in part, by nonoccupational factors. For bladder and kidney cancers, the excess risks are only suggestive of a true effect because of possible publication bias.

6.7.4 Genotoxicity of Lead

The NTP reviewed in some detail the genotoxicity studies over the period 1970-2002. These studies are cross-sectional studies, mostly of occupationally exposed workers compared to a control population. Usually blood lead levels are available to document exposure. Outcomes consisted of chromosomal aberrations (CA), sister chromatid exchange (SCE), micronuclei formation (MN), and studies of DNA damage (often via the comet assay) and/or measures of the mitotic activity. Of these outcomes, only CAs have been shown to have a positive relationship to subsequent cancer (Hagmar et al., 2004, Rossner et al., 2005). SCEs are generally considered a marker of exposure to environmental agents which have some effect on DNA, but are not thought to necessarily predict cancer risk. MN and DNA damage are thought to indicate genotoxicity with unknown effect on cancer risk. These outcomes are somewhat informative regarding the possible human carcinogenicity of lead but are clearly secondary to direct information on cancer risk from epidemiologic studies.

The most recent studies of the genotoxicity of lead are summarized in Annex Table AX6-7.1. Of eleven studies of chromosomal aberrations (CA), six were judged to show a positive relationship between CA and lead, four were judged negative, and one was neither clearly positive or negative. In general, these studies were done in the 1970s and 1980s; only one dates from the 1990s. There were nine studies of sister chromatid exchange. Of these, four were judged positive, three negative, and two could not be judged clearly one way or the other. It is notable that the positive studies were generally the most recent. There were four MN studies, all of which were judged positive. Finally, there were nine studies of DNA damage

and/or mitotic activity. These varied in the specific outcome, although many used a comet assay to measured oxidative damage to DNA. Eight of these nine studies were judged positive in the sense that increased DNA damage or mitotic activity was related to lead exposure, while one was judged negative.

Since the NTP review, there have been three additional cytogenetic studies which are informative regarding lead (Palus et al., 2003, Minozzo et al., 2004, and Fracasso et al., 2002), as well as one mutation study (Van Larebeke et al., 2004). All four of these studies (two of DNA damage one of MN, and one of a specific mutation frequency) were positive in significantly linking lead exposure to the outcome. The results of these studies as well as those reviewed by the NTP are summarized in Table 6-7.2.

Table 6-7.2. Results of Epidemiologic Studies on the Genotoxicity of Lead Exposure[a]

	Results		
Studied Outcome	**Positive**	**Mixed**	**Negative**
Chromosomal Aberrations (CA)	6	1	4
Sister Chromatid Exchange (SCE)	4	2	3
Micronucleus Formation (MN)	5	0	0
DNA Damage/Mitosis	10	0	1
Gene Mutation	1	0	0

[a] Results summarize the overall findings of epidemiologic studies addressing the potential genotoxic effects of lead exposure. Some studies addressed multiple aspects of genotoxicity; for these studies, their results for each of the listed categories of genotoxic outcomes are presented separately.

While the overall the evidence from cytogenetic studies is mixed, more recent studies which were focused on DNA damage or mitotic activity have tended to be largely positive. However, it is not known whether these outcomes predict subsequent cancer risk.

6.7.5 Review of Specific Studies on the Carcinogenicity of Lead Since the 1986 Lead AQCD

6.7.5.1 Introduction

The epidemiologic studies of lead exposure and cancer are listed in Table 6-7.3. The most relevant studies focus on exposure through occupational sources, wherein the most intense exposure to lead can be expected to occur. This exposure predominantly involves inorganic lead species. Relevant studies are discussed below, beginning with the most key studies of the general population will then be presented followed by a brief summary of other relevant studies examining the occupational studies.

6.7.5.2 Key Studies of Occupational Populations in the U.S.

There are seven key occupational studies based on highly exposed worker populations; these are all cohort studies with adequate numbers to address lung and/or stomach cancer. There are two cohorts based in the U.S. and five based outside it. Studies reviewed in this section are summarized in Annex Table AX6-7.2.

Steenland et al. (1992) followed up 1,990 male U.S. lead smelter workers, employed from 1940 to 1965, through 1988. Standardized mortality ratios indicated an excess of lung, stomach, kidney, and bladder cancer, but these excesses did not reach statistical significance. Focusing on a subgroup of workers classified as highly lead exposed based on air-monitoring records yielded a statistically significant excess for kidney cancer (standardized mortality ratio of 2.39 [95% CI: 1.03, 4.71]), although it did not appear to increase with increasing duration of exposure. Estimates for the other cancers (standardized mortality ratio of 1.11 [95% CI: 0.82, 1.47] for lung; 1.28 [95% CI: 0.61, 2.34] for stomach; 1.33 [95% CI: 0.48, 2.90] for bladder) showed little change with restriction to the high-exposure group. While neither arsenic nor cadmium exposure could be controlled for, 1975 NIOSH monitoring data indicated less intense exposure to airborne cadmium or arsenic than to lead. Lead averaged 3.1 mg/m^3 and arsenic 14 $\mu g/m^3$, compared to current OSHA standards of 0.05 mg/m^3 for lead and 10 $\mu g/m^3$ for arsenic. It is notable that a 1996 review of studies (Steenland et al., 1996) on arsenic-exposed workers concluded that significantly elevated rates of lung cancer were concentrated in studies where average exposures greatly exceeded OSHA standards (e.g., hundreds of $\mu g/m^3$). No data on workers' smoking status were available.

Table 6-7.3. Epidemiologic Studies of Lead Exposure and Cancer in Specific Populations, by Geographic Region and Study Design[a]

	Epidemiologic Study Design		
Specific Study Population	**Cohort**	**Nested Case-control**	**Case-control**
United States			
Battery and lead production workers	Cooper and Gaffey (1975), Cooper et al. (1985), Wong and Harris (2000)	Cooper et al. (1989), Wong and Harris (2000) (same publication as cohort study)	
Copper workers (Utah)	Rencher et al. (1977)		
Lead and zinc pigment plant workers	Sheffet et al. (1982)		
Lead smelter workers (Idaho)	Selevan et al. (1985), Steenland et al. (1992)		
Sample of deaths due to cancer vs. noncancer deaths (Illinois)			Mallin et al. (1989)
Brain cancer			Cocco et al. (1998a)
Central nervous system cancer			Cocco et al. (1998b)
Stomach cancer			Cocco et al. (1999)
NHANES II cohort mortality follow-up, general U.S. population	Jemal et al. (2002), Lustberg and Silbergeld (2002)		
Canada			
Population-based cases			Risch et al. (1988)
Specific cancers versus all cancers			Siemiatycki et al. (1991)
Europe			
Glass workers (Finland)	Sankila et al. (1990)		
Registry-derived liver cancer cases vs. stomach cancer or myocardial infarctions (Finland)		Kauppinen et al. (1992)	
Workers via Cancer Registry (Finland)	Anttila et al. (1995)	Anttila et al. (1996)	
Renal-cell cancer vs. population controls (Germany)			Pesch et al. (2000)
Laryngeal cancer among persons with no history of lead exposure (Greece)			Kandiloris et al. (1997)

Table 6-7.3 (cont'd). Epidemiologic Studies of Lead Exposure and Cancer in Specific Populations, by Geographic Region and Study Design[a]

Specific Study Population	Epidemiologic Study Design		
	Cohort	Nested Case-control	Case-control
Europe (cont'd)			
Glass workers (Italy)	Cordioli et al. (1987)		
Lead and zinc miners: females only (Sardinia)	Cocco et al. (1994b)		
Lead and zinc miners: male only (Sardinia)	Cocco et al. (1994a), Carta et al. (1994); Carta et al. (2003*)*		
Lead and zinc smelter workers (Sardinia)	Cocco et al. (1996)		
Lead and zinc smelter workers (Sardinia, but different from Cocco et al. 1996)	Cocco et al. (1997)		
Glass workers (Sweden)	Wingren and Englander (1990)	Wingren and Axelson (1985, 1987, 1993)	
Copper and lead smelter workers (Sweden)	Gerhardsson et al. (1995a)		
Copper and lead smelter workers (Sweden) (Lundström: full cohort; Englyst: sub-cohort)	Gerhardsson et al. (1986), Lundström et al. (1997), Englyst et al. (2001)		
Lead-acid battery workers (U.K.)	Dingwall-Fordyce and Lane (1963), Malcolm and Barnett (1982)		Fanning (1988)
Chromate (including lead-chromate) workers (U.K.)	Davies (1984a, 1984b)		
Zinc, cadmium, and lead smelter workers (U.K.)	Ades and Kazantzis (1988)	Ades and Kazantzis (1988) (same publication as cohort study)	
Asia			
Gliomas vs. noncancer patients (China)			Hu et al. (1998)
Meningiomas vs. noncancer patients (China)			Hu et al. (1999)
Gall bladder cancer vs. gallstone patients (India)			Shukla et al. (1998)
Prostate cancer cases versus benign prostate hyperplasia cases and normal controls (India)			Siddiqui et al. (2002)

[a] Within regions, study populations are listed in chronological order based on the earliest published study on that specific worker population. Publications considered to be key studies are italicized.

Wong and Harris (2002) extended follow-up on the battery and smelter worker cohort previously reported on by Cooper et al., 1985 through 1995, an additional 15 years. With the additional follow-up, standardized mortality ratios for lung, tracheal, or bronchial cancer decreased to 1.14 (95% CI: 0.99, 1.30) for battery workers but showed little change for smelter workers at 1.22 (95% CI: 1.00, 1.47). A significantly elevated standardized mortality ratio for stomach cancer (1.53 [95% CI: 1.12, 2.05]) persisted among battery workers, with a lesser elevation among smelter workers (1.33 [95% CI: 0.75, 2.20]). Among other cancers, only thyroid cancer among all workers combined showed a significantly elevated standardized mortality ratio (3.08 [95% CI: 1.33, 6.07]). Cancer mortality did not increase with earlier year of hire for lung, stomach, or thyroid cancer. Lung and stomach cancer mortality peaked among workers with 10 to 19 years of factory employment and declined with longer employment duration. Thyroid cancer mortality occurred exclusively among workers with 20 or more years of exposure. As with earlier analyses based on this cohort, concomitant exposures to other compounds could not be controlled for, but as these were likely to be most intense among lead production workers, whose standardized mortality ratios were similar to or lower than those for battery workers, any bias resulting from such exposure probably was toward the null. No data were available to assess the possible role of smoking, diet, or other potential nonoccupational risk factors in the results.

A nested case-control analysis was also conducted to further explore stomach cancer mortality within workers employed at the Philadelphia lead battery plant in the cohort (Wong and Harris, 2000). Among 30 workers who died of stomach cancer and 120 age-matched controls, job title histories were used to estimate duration of employment and cumulative exposure based on job-specific intensities of exposure. Duration of employment and estimated degree of lead exposure showed no elevation among workers who died of stomach cancer, nor did mortality increase across increasing tertiles of lead exposure. Little information appeared to be available on potential confounders. The authors suggested that in light of historically higher stomach cancer rates in Ireland and Italy and the observation of a higher proportion of Irish and Italian immigrants among lung cancer cases in the case-control study, differences in ethnicity may have contributed to the elevated standardized mortality ratios seen in the cohort as a whole. The recent IARC Working Group (IARC, 2005) concluded that, based on the ethnic composition

of the control population (23% Irish or Italian), confounding by race could account for only part of the observed association, however.

The extended follow-up and nested case-control analyses on the original Cooper et al. (1985) cohort thus continued to provide evidence for some increase in lung and stomach cancer among these lead workers, but no consistent evidence of increasing cancer risk with increasing exposure within the lead worker cohort itself, especially for stomach cancer.

Fanning (1998) studied deaths due to specific cancer types among U.K. battery and other factory workers. High to moderate lead exposure resulted in odds ratios for lung and digestive cancer of 0.93 and 1.13, respectively, with the latter elevation due mainly to stomach cancer (odds ratio of 1.34). No odds ratios reached nominal statistical significance, and no associations were noted for other cancer types. The excess of digestive cancer deaths was restricted to the 1926 to 1965 period, during which lead exposures would have been most intense. Odd ratios for other cancers did not vary by period. Because each cancer case group was compared with a control group consisting of subjects who died from all other causes, including other cancers, odds ratios would have been biased downward if some of these other deaths also were lead-related. However, most deaths were due to nonmalignant respiratory or circulatory diseases other than hypertension, mitigating the potential impact of such a bias.

Anttila et al. (1995) linked 20,700 Finnish workers whose blood lead was monitored during 1973 to 1983 by the Finnish Institute of Occupational Health to the Finnish Cancer Registry. Exposure was subdivided according to highest peak blood level measured: low (0 to 0.9 μmol/L [0 to 18.6 μg/dL]), moderate (1.0 to 1.9 μmol/L [20.7 to 39.4 μg/dL]), and high (2.0 to 7.8 μmol/L [41.4 to 161.6 μg/dL]). The total cohort showed no elevation in total or site-specific cancer mortality, based on an standardized mortality ratio analysis. Among male workers with moderate exposure, incidence of total respiratory cancer and lung cancer both were elevated (standardized incidence ratio of 1.4 [95% CI: 1.0, 1.9 for both]). Risks of total digestive, stomach, bladder, and nervous-system cancer also were modestly elevated. However, risks did not increase in the high-exposure group. Risks of mortality for all cancer for both men and women (relative risk 1.4 of [95% CI: 1.1, 1.8]) and lung or tracheal cancer (relative risk of 2.0 [95% CI: 1.2, 3.2]) were even stronger when a person-year analysis was applied to compare workers with moderate lead exposure to those with low exposure. Again, risks did not increase in the highest exposure group. This exposure group was smaller than the others, however, which

limited the power of analyses specific for high exposure workers. Thus, for example, the numbers of lung or tracheal cancer deaths among men in the low-, moderate-, and high-exposure groups were 25, 34, and 11, respectively, for the person-year-based analyses.

It should be noted that for cancer, cumulative exposure, particularly during the earlier part of the follow-up period, might be more relevant than peak exposure, although the two were reported to be highly correlated (Anttila et al., 1995). Case-referent substudies of lung cancer used different exposure criteria (Anttila et al., 1995). Odds ratios increased most consistently with increasing cumulative exposure to lead. Among histologic subtypes, significantly elevated risk for squamous-cell cancer of the lung (odds ratio of 4.1 [95% CI: 1.1, 15]) for the highest blood lead group persisted after adjustment for smoking, although with additional adjustment for engine exhaust and solvent exposure, the risk declined (odds ratio of 3.4 [95% CI: 0.9, 13]). Results for female workers are not considered, as too few cancers (3 total) occurred to permit meaningful conclusions. Although the follow-up period was relatively short, the lung cancer association was analyzed in much greater detail than in most studies, and smoking was adjusted for, while the association between lead exposure and lung cancer weakened with control for engine exhaust and solvent exposure, the odds ratio remained well above 1. The highest odds ratio of all was observed for estimated risk of lung cancer among workers with peak blood lead levels of at least 0.8 μmol/L [≥16.6 μg/dL] who were exposed to engine exhaust (odds ratio of 14.9 [95% CI: 1.3, 178]; 11 cases). If engine exhaust was acting as an effect modifier, directly controlling for it might not have been appropriate. The exhaust could have served as a source of organic lead, as well.

Gerhardsson et al. (1995a) followed up 664 male Swedish secondary lead smelter workers, tracing their cancer morbidity from 1969 to 1989. Compared to the population of the surrounding county, the workers' standardized incidence ratio for all cancers was 1.27 (95% CI: 0.91, 1.74), based on 40 tumors. Standardized incidence ratios for cancers at all specific sites except the brain were elevated, notably those for the respiratory system (1.32 [95% CI: 0.49, 2.88]), stomach (1.88 [95% CI: 0.39, 5.50]), and colon (1.46 [95% CI: 0.30, 4.28]). Because of the small numbers of tumors (only 6, 3, and 3, respectively, even for the aforementioned sites), the reliability of estimates for most sites is limited. Restricting analyses to workers in the highest quartile of lead exposure based on routine blood lead monitoring data yielded a higher standardized incidence ratio for total gastrointestinal cancer (2.43 [95% CI: 1.11, 4.62];

9 tumors), but not respiratory cancer. Availability of blood lead measurements is an advantage of this study, along with a lead-exposed worker population unlikely to have much exposure to arsenic, chromium, or cadmium. However, the cases were too few for detailed exposure-response analyses by cancer type. Lack of data on smoking further restricts interpretation of the results.

Lundström et al. (1997) followed 3,979 Swedish smelter workers from 1928 to 1987. Workers were further subdivided into those with high cumulative blood lead scores (mean times years exposed >10 μmol/L), and those exposed to "lead only" (excluding those from departments thought to have significant exposures to other potential carcinogens, such as arsenic, or little exposure to lead). The lung cancer standardized mortality ratio was 2.8 (95% CI: 2.0, 3.8) for the total cohort, 2.8 (95% CI: 1.8, 4.5) for the high-exposure subgroup, and reportedly similar for the subgroup exposed to lead only. With adjustment for a 15-year latency period, lung cancer standardized incidence ratios likewise differed little between the total cohort and high-exposure subgroup; however, among workers with exposure to lead only, the standardized incidence ratio rose from 3.1 (95% CI: 1.7, 5.2; 14 cases) for all workers to 5.1 (95% CI: 2.0, 10.5; 7 cases) for those with the highest exposure. With a 15-year latency period, elevated standardized incidence ratios also were observed for cancer of the brain and nervous system (1.6 [95% CI: 0.4, 4.2]) and renal pelvis, ureter, or bladder (1.8 [95% CI: 0.8, 3.4]) among the high-exposure subgroup. Non-respiratory cancers were too infrequent (5 total) in the high-exposure lead-only subgroup for meaningful analysis. This study's size, extensive follow-up, and ability to integrate blood-based and job-based exposure indices give it unusual power. The apparent increase in cancer risk with higher cumulative lead exposure that appeared when workers thought to be potentially exposed to other metals, such as arsenic and nickel, were excluded also appeared to strengthen the evidence for a specific link between lead and respiratory cancer. A subsequent study by Englyst et al. (2001), however, cast doubt on the efficacy of the "lead only" grouping.

Englyst et al. (2001) conducted additional analyses on one element of the Lundström et al. (1997) cohort. A total of 1,093 workers from the smelter's lead department was followed up through 1997. Significantly elevated lung cancer standardized incidence ratios were observed in all subcohorts, including the subcohort who had never worked in arsenic-exposed areas (3.6 [95% CI: 1.2, 8.3]; 5 cases). This subcohort is the same as the "lead-only" subgroup evaluated by Lundström et al. (1997). A review of detailed job histories obtained for all workers

with lung cancer, however, indicated that 13 of the 15 had "considerable" exposure to arsenic as well as lead, including all but 1 in the "lead only" subcohort.

Carta et al. (2003) followed up the mortality of 918 Sardinian lead smelter workers from 1972 through 2001. Smelter workers as a whole displayed an overall cancer mortality no higher than expected based on regional rates (standardized mortality ratio of 1.01). Cancer-specific standardized mortality ratios were, however, nonsignificantly elevated for cancers of the lung (1.21) and stomach (1.22) as well as for lymphoma and leukemia (1.82). Use of blood and ambient lead monitoring data available by department and task to categorize estimated exposure yielded a statistically significant upward trend with increasing lead exposure for lung cancer; no significant trend was seen for the other cancers, although in light of the small number of gastric cancer and lymphoma/leukemia deaths (4 and 6, respectively) interpretation of dose-response is problematic for these outcomes.

6.7.5.3 Key Studies of the General Population

There are two key general population cohort studies in which lead exposure is assessed via blood lead levels (see Annex Table AX6-7.3 for additional details). Jemal et al. (2002) conducted the first biomarker-based general population cohort study of lead exposure and cancer. The study employed the subsample of 3,592 white U.S. participants in NHANES II (1976 to 1980) who had undergone blood lead level determinations at time of entry. Deaths among this population were enumerated through 1992 by linkage to the National Death Index (NDI) and Social Security Administration Death Master File. Median blood lead levels in this population were 12 μg/dL. Adjusted for age, smoking, drinking, region, year, and gender, risk of mortality from any cancer rose across quartiles of blood lead level, but this trend was not statistically significant. The trend across quartiles was not consistent in gender-specific analyses, although relative risks were elevated for the highest quartile of blood lead level in both men and women (relative risk 2.0 for men and 1.6 for women). The relative risk for lung cancer based on comparison of subjects with blood lead levels above or below the median was 1.5 in the combined population, with higher risk observed among women (2.5 [95% CI: 0.7, 8.4]) than men. The highest relative risks were observed for cancer of the esophagus (3.7 [95% CI: 0.2, 89]), pancreas (3.6 [95% CI: 0.6, 19.8]), and stomach (2.4 [95% CI: 0.3, 19.1]); no elevations were noted for cancers of other sites.

The lack of statistically significant results reflects the small number of deaths during follow-up, which limited the study's power; of the nine major sites examined, the number of deaths ranged between 5 and 16 for all sites except the lung. Detailed exposure-response analyses were restricted to all cancers combined, although potential effects could have been strongly target-organ specific. In addition, the use of quartile cut points based on the distribution of lead concentrations estimated for the total U.S. population resulted in relatively small numbers in the referent group (lowest exposure quartile) for males and in the high-exposure quartile for females. Use of a biomarker provided an objective measure of lead exposure. Nevertheless, reliance on a single blood lead measurement produces less reliable estimates than would be obtained through multiple measurements and precludes addressing temporal changes in lead exposure over the follow-up period. Lack of control for exposure to occupational carcinogens other than lead and potential residual confounding by duration and intensity of tobacco smoking also could have biased the results, especially for men. Lustberg and Silbergeld (2002) carried out another biomarker-based general population study based on the same NHANES II mortality cohort used by Jemal et al. (2002). This study did not exclude nonwhites, however (thus gaining 524 subjects) and employed more extensive adjustment for potential confounding factors than the Jemal et al. (2002) analyses (i.e., education, body mass index, and exercise were included in the regression models, although alcohol intake was not). In addition, persons with blood lead levels of 30 μg/dL or higher were excluded in order to restrict comparisons to levels below the OSHA standard for lead exposure. Persons with levels below 10 μg/dL served as the referent group. Survival analyses adjusted for potential confounders found a relative risk for cancer mortality of 1.5 (95% CI: 0.9, 2.5) for those with blood lead levels of 10 to 19 μg/dL, compared with those with levels below 10 μg/dL, rising to 1.7 (95% CI: 1.0, 2.8) for those with levels of 20 to 29 μg/dL. Separate analyses of lung-cancer and non-lung-cancer deaths yielded estimates of increased risk for moderate- or high-exposure groups, compared with the referent population, both for lung cancer and non-lung cancer. However, none of the estimates reached the $P < 0.05$ level of statistical significance, and the results for non-lung cancers showed no evidence of an exposure-response relationship.

As with Jemal et al. (2002), the use of a biomarker for exposure and the prospective design of the study are strengths. Its attempts to control for potential confounders were more extensive, and its choice of cut points for the referent category yielded more males in the referent

group, although that group still included less than 20% of the study population. However, it is notable that blood lead levels rose significantly with smoking level. The models included terms for former smoking, current light smoking, and current heavy smoking (>1 pack per day). Nevertheless, some degree of residual confounding due to smoking might have remained, which could have contributed to the estimated risk of lung cancer for the highest exposure category (relative risk of 2.2 [95% CI: 0.8, 6.1]). Such residual confounding would have had less effect on the results for non-lung cancer. As noted regarding the other NHANES-based study, however, mortality due to cancers of other sites was too uncommon to allow for reliable site-specific comparisons. In the Lustberg and Silbergeld analysis, all cause and cardiovascular mortality increased monotonically with blood lead level, which might indicate residual confounding from SES or smoking affecting both heart disease and cancer.

6.7.5.4 Other Lead Studies

There are a variety of other epidemiologic studies of lead exposure, which are less important than the key studies above but which offer some information. Studies reviewed in this section are summarized in Annex Table AX6-7.4. Rencher et al. (1977) compared Utah copper smelter workers' mortality with that of miners for the same company. Workers in lead-exposed operations had a higher proportional mortality due to respiratory cancer in general and lung cancer specifically than did other workers, with or without control for smoking status. Among lead-exposed workers, those who developed lung cancer had significantly higher estimated lead exposure than the rest. Workers with lung cancer also had significantly higher estimated exposure to arsenic and sulfur dioxide, however, and these exposures were not adjusted for.

Ades and Kazantzis (1988) conducted a cohort study of lung cancer mortality among 4,393 U.K. zinc, lead, and cadmium smelter workers. Smelter workers had a lung cancer standardized mortality ratio of 1.25 (95% CI: 1.07, 1.44) compared with national rates, based on 182 lung cancer deaths Potential effects of lead could not be adjusted for arsenic exposure or other exposures due to inadequate numbers. Cancer-specific standardized mortality ratios were calculated for production and maintenance workers from an Italian lead and zinc smelter followed from 1950 to 1992 by Cocco et al. (1997). Deaths from lung,, stomach, and all cancer were not elevated over regional rates. Cocco et al. (1996) followed 1,060 Sardinian lead and zinc smelter workers whose glucose-6-phosphate dehydrogenase (G6PD) phenotype had been

measured from 1973 through 1991. Despite the thought that G6PD-deficient workers might be more vulnerable to the depletion of red blood cell glutathione associated with lead toxicity, mortality from cancer and from all causes was slightly lower among G6PD-deficient workers than among G6PD-normal workers. Follow-up was subsequently extended through 2001 by Carta et al. (2003).

Three European studies followed up cohorts of glass workers. Cordioli et al. (1987) studied 468 Italian glass workers. Workers producing low-quality glass containers were classified as being exposed to lead. A small elevation in mortality from all cancer (standardized mortality ratio of 1.3 [95% CI: 0.8, 1.8]) among glass workers was driven by significant excesses in lung cancer (2.1 [95% CI: 1.1, 3.6]) and laryngeal cancer (4.5 [95% CI: 1.2, 11.4]). The small number of deaths among exposed workers (28 total, 13 lung, and 4 laryngeal cancer) limited the study's statistical power. Sankila et al. (1990) compared the incidence of cancer in 1,803 male and 1,946 female Finnish glass workers with that of the national population. Glassblowers were considered to be a lead-exposed subgroup. Modest elevations in lung cancer risk were observed among glass workers for both men (standardized incidence ratio of 1.3 [95% CI: 1.0, 1.7]) and women (1.1 [95% CI: 0.5, 2.3]). However, the increased risk of lung cancer was not specific to glassblowers. In the final study, Wingren and Englander (1990) compared mortality in Swedish glass workers from work areas with airborne lead levels ranging from <0.001 up to 0.110 mg of lead/m^3, noting a significant elevation for pharyngeal cancer (standardized mortality ratio of 9.9 [95% CI: 1.2, 36.1]) and nonsignificant elevations for lung and colon cancer compared to national rates.

Wingren and Axelson (1985, 1987, 1993) conducted a case-control analysis comparing stomach, colon, and lung cancer mortality among Swedish glass workers with that of the surrounding regional populations. A small early study of three parishes (Wingren and Axelson, 1985) was expanded to include 11 parishes, thus encompassing most of the Swedish glass-work industry (Wingren and Axelson 1987). Mortality from cancer of the lung (odds ratio of 1.7 [90% CI: 1.1, 2.5]), stomach (1.5 [90% CI: 1.1, 2.0]), and colon (1.6 [90% CI: 1.0, 2.5]) all were elevated among glass workers as a whole (Wingren and Axelson, 1987). Among specific classes of glass workers, glassblowers had the highest odds ratios (2.3, 2.6, and 3.1 for lung, stomach, and colon cancer, respectively). When the data were analyzed according to level of estimated metal exposure, no consistent dose-response trend with lead was found for lung cancer, and the

association with stomach cancer was weaker for lead than for arsenic, copper, and other metals. In general in this study it was difficult to separate the independent cancer effects of different metals.

Sardinian lead and zinc miners were studied in a set of three papers published in 1994. Carta et al. (1994) studied a small group of workers and Cocco et al. (1994a,b) expanded coverage to follow 1,741 male and 526 female workers from two mines. Number of cancer deaths were small. This study was limited because exposure characterization focused only on silica and radon daughters; no lead exposure specific analyses were performed.

Davies (1984b) followed up 57 pigment factory workers who had been diagnosed with nonfatal lead poisoning, finding a small excess of lung cancer deaths (relative risk of 1.45), but with only 4 deaths in the lead-poisoned group this result did not reach statistical significance.

Mallin et al. (1989) used death certificates for Illinois males to compare deaths from seven specific cancers with a control group of 3,198 randomly selected deaths from other causes. Based on occupations from death certificates, the odds ratio for cancer of the brain (3.0, $p < 0.05$) was significantly elevated in white male glass workers (as well as physicians and communications workers). No significant association was observed for other cancer sites, including lung and stomach. This isolated association is not consistent with the results for Swedish glass workers summarized above. The National Cancer Institute, NIOSH, and the National Center for Health Statistics have assembled a database that integrates industry, occupation, and cause of death information from death certificates in 24 states. This resource provides a very large sample size for case-control analyses of occupational exposures, but results are limited by a lack of detailed work history and no control over confounders. Cocco et al. (1998a) matched all 27,060 brain cancer deaths occurring among persons aged 35 or older during 1984 to 1992 with four gender-, race-, age-, and region-matched deaths from nonmalignant causes. A job-exposure matrix was used to assign subjects to low, medium, or high probability and intensity of exposure. Risk of brain cancer mortality increased consistently with rising intensity of lead exposure among African American men but not among the other three race-gender groups. Cooco et al. (1998b) broadened the study to CNS cancer deaths, and computed odds ratios for specific industries and occupations rather than particular substances. Statistically significant associations were found with some industries and some race/sex groups, but little inference can be made about lead carcinogenicity from these data.

In the third 24-state death-certificate study, 41,957 stomach cancer deaths were matched with 83,914 deaths due to nonmalignant causes (Cocco et al., 1999). A job-exposure matrix was used to assign subjects to low, medium, or high probability and intensity of exposure to lead and 11 other chemicals. Elevated odds ratios occurred among white women (1.53 [95% CI: 1.10, 2.12]), African-American men (1.15 [95% CI: 1.01, 1.32]), and African-American women (1.76 [95% CI: 0.74, 4.16]) with high probability of lead exposure. Odds ratio in the moderate-probability group were elevated only for African-American women (1.37 [95% CI: 0.58, 3.21]), and not elevated for any exposure group among white males. Risk showed no consistent increase with intensity of exposure in any group. The absence of any association with lead exposure among the largest race-gender group, white males, is notable, as is the general absence of association with intensity of exposure. More consistent elevations of odds ratios for stomach-cancer mortality were observed for inorganic dust and nitrosamines than for lead.

Anttila et al. (1996) presented a nested case-control analysis of 26 Finnish male workers with central nervous system (CNS) cancer and 200 controls, using the same Finnish occupational cohort as in Anttila et al. (1995). For CNS cancer incidence, odds ratios rose with increasing peak lifetime blood lead level; however, the trend was not statistically significant. Odds ratios for glioma mortality rose consistently and significantly with increasing peak and mean blood lead level, as well as duration of and estimated cumulative lead exposure. A strength of this study is the availability of blood lead measurements. Limitations include the small number of cases (10 gliomas among workers with complete exposure information), short follow-up time (maximum of 15 years), potential selection bias due to low response rates (60% for cases, 56% for controls), and possible coexposures such as solvents or other metals.

Risch et al. (1988) compared 826 Canadian men with histologically confirmed bladder cancer with 792 Canadian population controls. Reported occupational exposure to lead yielded a significantly elevated smoking-adjusted odds ratio (2.0 [95% CI: 1.2, 3.5]) and a significant trend with duration of exposure. Of 17 other exposures examined, only one (tar and asphalt) was significantly associated with bladder cancer. These analyses relied on self-reported exposure, with the potential for inaccurate recall. Siemiatycki et al. (1991) conducted a case-control study in Canada using 3,730 cases of various histologically confirmed cancers. Occupational exposure to 293 substances, including lead, was estimated from interview data. Elevated odds ratios were noted for cancer of the lung (1.1 [90% CI: 0.9, 1.4]), stomach (1.2 [90% CI: 1.0, 1.6]), bladder

(1.3 [90% CI: 1.0, 1.6]), and kidney (1.2 [90% CI: 1.0, 1.6]). Strengths of this study are adjustment for smoking and other potential risk factors and reliance on interview-obtained exposure data, further evaluated by experts. Limitations include potential confounding by the other 292 occupational exposures and low quantitative detail regarding lead exposure.

Kauppinen et al. (1992) conducted a nested case-control study in Finland, matching 344 primary liver cancer deaths by age and gender to 476 stomach cancer deaths and 385 myocardial infarct deaths. No association was found between lead and liver cancer, which was not an a priori site of interest. Use of a control group with stomach cancer, which some other studies have linked to lead exposure, may have biased results toward a negative association.

In a Chinese hospital-based case-control study, Hu et al. (1998) compared 218 patients with histologically confirmed primary gliomas with 436 patients with non-neurological, nonmalignant disease, matched by age, gender, and residence. An odds ratio could not be calculated for occupational exposure to lead because no glioma patients reported such exposure.

In a parallel study, Hu et al. (1999) compared 183 patients with histologically confirmed primary meningiomas with patients with non-neurological, nonmalignant disease, matched by age, gender, and residence. Reported occupational exposure to lead was associated with risk of meningioma in both men (odds ratio of 7.20 [95% CI: 1.00, 51.72]) and women (5.69 [95% CI: 1.39, 23.39]). Some elevation of odds ratios occurred in most of the 14 occupational exposures examined, including exposure to cadmium. Malczyk et al. (1999) measured urinary lead concentrations in 24 Polish bladder cancer cases. Ten out of the 24 cases had urinary lead levels above 90 μg/L, thus exceeding the upper limit of the range estimated as normal for a healthy person (10-90 μg/L). Results are limited by the lack of any measurements done on persons without bladder cancer from the same area.

Pesch et al. (2000) compared occupational exposure to potential carcinogens among 935 Germans newly diagnosed with renal-cell cancer and 4,298 controls selected from regional population registries and matched by age, gender, and area of residence. Lifetime job histories and information on smoking habits and other potential risk factors were collected by interview. Cumulative exposure to lead, as well as cadmium, solder fumes, welding fumes, and metals in general, was estimated based on previously published job exposure matrices and grouped into four ascending categories; separate estimates of lead exposure were calculated based on British- and German-developed matrices. After adjustment for age and smoking, odds ratios for renal

cancer were elevated in men (1.5 [95% CI: 1.0, 2.3]) and women (2.6 [95% CI: 1.2, 5.5]) with the highest lead exposure, compared with the low-exposure groups based on the British matrix. When exposure was based on the German matrix, the odds ratio was less elevated among men (1.3 [95% CI: 0.9, 2.0]); no results for women were reported. Strengths of the study are its size and population base. The primary limitation is uncertainty regarding the specificity of the results for lead. Significant associations also were noted for exposure to cadmium, solder fumes, and organic solvents among men, for example, but no analyses attempting to account for other exposures were reported. It is thus unclear how much of the observed risk associated with lead exposure may be secondary to exposure to cadmium or other agents.

Siddiqui et al. (2002) compared blood lead levels in Indian men with prostate cancer or benign prostatic hyperplasia to levels seen in normal controls of similar SES. Lead levels were significantly higher in both prostate cancer and benign prostatic hyperplasia cases than in controls, while zinc levels were lower.

Kandiloris et al. (1997) found similar blood lead levels but lower aminolevulinic acid dehydratase (ALAD) activity in 26 laryngeal carcinoma cases compared to 53 controls from the same hospital. Shukla et al. (1998) found significantly higher mean bile lead in 38 newly diagnosed, histologically confirmed gall bladder cancer cases compared to 58 patients with gallstones diagnosed at the same Indian hospital surgical unit (58.38 ± 1.76 mg/L versus 3.99 ± 0.43 mg/L). Cancer cases also showed elevated cadmium and chromium levels.

6.7.6 Confounding of Occupational Lead Studies Due to Other Occupational Exposures: Arsenic, Cadmium

A number of studies of lead workers come from smelters, where exposures to other metals are common. Of particular concern are other lung carcinogens, especially arsenic (workers exposed to high levels of arsenic historically have had a lung cancer relative risk of 3-4, see Steenland et al. 1996), but also cadmium. Glass workers are also of limited use for inference about lead effects, as they are also typically exposed to arsenic, cadmium, chromium, and nickel, all of which are lung carcinogens (e.g., see Wingren and Axelson, 1993).

In some smelters, measurements have been taken which indicate clearly that exposures to these other carcinogens was minimal and the main suspect is lead (e.g., Steenland et al., 1992). In others, however, one is unable to disentangle the effects of arsenic and lead (Ades and

Kazantis, 1988, Lundstrom et al., 1997). As a result, these studies cannot yield strong evidence regarding the possible relation between lung cancer and lead specifically. The study by Lundstrom et al., 1997 is particularly important in this regard, because it had a high relative risk of 2.8 (95% CI: 2.0, 3.8), and had an important effect in raising the overall result when included in meta-analyses (e.g., Steenland and Boffetta [2000], where exclusion of the Lundstrom et al. study lowered the estimated combined lung cancer relative risk from 1.30 to 1.14). A subsequent publication by Englyst et al. (2001) indicated that the smelter workers studied by Lundstrom et al. (1997) were likely to have had significant exposure to arsenic, and the authors concluded that it was impossible to separate the effects of lead and arsenic.

6.7.7 Confounding of Lead Studies: Smoking and Other Factors

The most informative studies of lead carcinogenicity are those comparing highly exposed workers to general populations. In these comparisons one must consider typical differences between worker populations and the general populations, in particular differences due to smoking and diet. Smoking can be a major confounder for lung cancer, while diet or SES can be a confounder, albeit weaker, for stomach cancer.

Regarding smoking, it has been shown both theoretically and empirically that confounding due to smoking differences between workers and the general population will typically account for an observed relative risk of approximately 1.1 to 1.2, with a possible maximum of about 1.4 (Axelson and Steenland, 1988; Siemiatycki et al., 1988). Furthermore, most occupational cohort studies are retrospective and have little information on smoking, making it impossible to control directly for potential confounding by this strong risk factor. As noted above, the lung cancer relative risk in the meta-analysis of Steenland and Boffetta (2000), after excluding the Lundstrom et al. study, was 1.14 (95% CI: 1.04, 1.73), based on seven occupational cohort studies, six of which used a non-worker external referent population, and none of which controlled for smoking as a confounder. This relatively small excess relative risk could plausibly be due to confounding by smoking. Unfortunately the occupational cohort studies were usually not followed by nested-case control studies of lung cancer which could have controlled for smoking, and furthermore they usually did not involve internal exposure-response analyses, wherein confounding by smoking is usually minimal. An exception was the lung cancer case-control study conducted by Anttila et al. (1995) within a large cohort of Finnish

workers with known blood lead levels. In this case-control study smoking-adjusted lung cancer odds ratios were increased among workers with higher estimated cumulative blood lead or higher peak blood lead exposure compared to workers with the lowest exposure, and the authors noted that smoking actually appeared to be a "weak negative confounder" for the high peak blood lead group. Also, in one large population-based case-control study with extensive information on other cancer risk factors, there remained an elevated odds ratio for lung cancer with substantial lead exposure after controlling for smoking (Siemiatycki et al., 1991). Hence there is some evidence that confounding by smoking does not explain the modest excess lung cancer risk seen in many studies.

Diet high in salt or smoked meats, *Helicobacter pylori* infection, and SES are possible confounders for stomach cancer. Those of highest SES compared to those of lower SES have been shown to have a relative risk of about 3 (Tomatis, 1990). None of the occupational cohort studies, in which again stomach cancer in workers was compared to the general population, controlled for these potential confounders. However, these potential confounding factors are much less powerful risk factors in respect to stomach cancer than smoking is with respect to lung cancer, and hence are unlikely to account for relative risks higher than perhaps 1.1 or at most 1.2. Given that the occupational cohort studies had a combined relative risk of 1.34 (95% CI: 1.14, 1.57) in the meta-analysis of Steenland et al. (2002) and 1.33 (95% CI: 1.18, 1.49) in that of Fu and Boffetta (1995), it seems unlikely that confounding by these factors can fully account for the excess stomach cancer risk observed in the occupational studies.

6.7.8 Summary of Epidemiologic Evidence for the Genotoxic and Carcinogenic Effects of Lead

The availability of studies of cancer in lead-exposed populations was relatively limited at the time of the 1986 Lead AQCD. The number and range of studies has notably expanded since that time, including extended follow-ups of major extant cohorts, new cohort and case-control studies, and analyses addressing not only cancer but genotoxicty. These new human data greatly expand our knowledge of possible lead carcinogenicity. Animals studies are primarily based on dermal exposure to lead acetate. While the animal studies clearly show a carcinogenic effect, they are of less relevance here because human exposures are usually to inhaled lead oxides.

Studies of genotoxicity consistently link lead exposed populations with DNA damage and micronuclei formation, although less consistently with the more established indicator of cancer risk, chromosomal aberrations. Epidemiologic studies, particularly those of the high exposed occupational cohorts, are the most informative for determining whether lead causes cancer, because in general we assume that any cancer effect will be strongest and most easily detected when exposure is highest. There are only two general population cohort studies at ambient levels, and these are of the same population (NHANES II in the late 1970s). These general population studies at lower exposure levels show internal dose-response trends but suffer at times from small numbers for site-specific analyses or lack of site-specific analyses altogether and, also, from possible residual confounding by SES and smoking.

The strongest evidence in the key occupational studies linking lead exposure to actual human cancers is that for cancers of the lung and those of the stomach. Of seven large occupational cohort studies available (Ades and Kazantzis, 1988; Anttila et al., 1995; Carta et al., 2005; Gerhardsson et al., 1995; Lundstrom et al., 1997; Steenland et al., 1992; Wong and Harris, 2000), for example, all showed results consistent with an increase in lung cancer risk among lead-exposed workers, and in four of these studies the association was statistically significant. Further, where workers could be categorized as to their level of lead exposure, the greatest magnitude of association for lung cancer was usually seen for the highest exposure category. However, the modest elevation of lung cancer risk seen in most relevant studies is in the range of possible confounding due to smoking or other occupational exposures, particularly arsenic, which precludes the evidence from these studies being seen as conclusive. In particular, the one occupational study with the highest lung cancer risk (Lundstrom et al.) has been subsequently shown to be highly confounded by arsenic, and without this study, the combined evidence for a lung cancer elevation across studies is considerably reduced (e.g., the estimated relative risk falls from 1.30 to 1.14). A moderate elevation of stomach cancer is also found in most studies of occupationally exposed populations with applicable data on this outcome. As with lung cancer, it is possible that other risk factors such as intake of smoked meats or *H. pylori* infection could have contributed to the observed associations, but the observed elevation (meta-analysis of 1.33 or 1.34) coupled with the known effect of diet makes it unlikely that the elevation in stomach cancer is entirely due to confounding by diet . Data for other sites such as kidney, brain, and

bladder show some indications of an excess, but the results across studies are not consistent and are based on small numbers.

The epidemiologic data reviewed above from key high lead exposure occupational studies suggest a relationship between lead exposure and cancers of the lung and the stomach. These are supported by two meta-analyses. This is limited by potential confounders such as other occupational exposures (arsenic, cadmium), smoking, and dietary habits. General population cohort studies in which low lead exposure was assessed via blood levels and adjusted for confounders showed trends for a relationship, but were limited by relatively small numbers for site-specific analysis. A cancer assessment on lead has not been conducted using the U.S. EPA Guidelines for Cancinogen Risk Assessment (U.S. Environmental Protection Agency, 2005). However, the most recent IARC (2005) review concluded that inorganic lead compounds were probable human carcinogens (Group IIA), based on limited evidence in humans and sufficient evidence in animals. This classification is one step down from a classification as "definite" human carcinogen (Group I).

6.8 EFFECTS OF LEAD ON THE IMMUNE SYSTEM

6.8.1 Summary of Key Findings of the Effects of Lead on the Immune System from the 1986 Lead AQCD

The 1986 Lead AQCD concluded that studies conducted in laboratory animal models provided evidence for immunosuppressive effects of lead; however, evidence for such effects in humans was lacking. Since then, the epidemiological study of immunological effects of lead has progressed considerably. The currently available epidemiologic and clinical observations are consistent with the greater body of evidence derived from studies in experimental animals indicating that lead can suppress cellular and humor immunity and decrease host resistance to infection agents and tumor cells (see Section 5.9). Findings from the epidemiologic studies suggest that lead exposure (as reflected in blood lead concentration) may be associated with effects on cellular and humoral immunity. These effects include changes in serum immunoglobulin levels (e.g., elevated serum IgE); perturbation of peripheral lymphocyte phenotype profiles, including decreases in peripheral blood T-cell abundance and changes in

T-cell:B-cell abundance ratios; suppression of lymphocyte activation; and suppression of neutrophil chemotaxis and phagocytosis.

Available studies of associations between lead exposure and immunological outcomes are summarized in Annex Tables AX6-8.1 and AX6-8.2. In general, while the studies provide support for associations between lead exposure and immunological outcomes, the studies have numerous limitations that complicate the assessment of the strength of the associations and causation. Furthermore, the health consequences of outcomes that have been associated with lead exposure are uncertain. All studies have been cross-sectional in design and most included relatively small cohorts. The studies implemented varying degrees of quantitative analysis of potential covariables and confounders. In most studies, a detailed analysis of covariables and confounding was lacking, and many of the reports offered no analysis of covariables or confounding. Covariables that were considered (but not consistently) in multivariate analyses or controlled by stratification included age, sex, race, smoking habits, alcohol consumption, and illness and/or medications that might affect the immune system. Studies that offer the strongest designs are discussed in greater detail below.

6.8.2 Host Resistance

Associations between lead exposure and host resistance have not been rigorously examined in humans. Two analyses of illness surveys in children (Rabinowitz et al., 1990) and lead workers (Ewers et al., 1982) have been reported, which suggest a possible association between increasing blood lead concentrations (>10 μg/dL) and illness incidence or prevalence. Both studies relied on personal surveys for assessment of illness and neither study considered covariates or confounders in the analyses.

6.8.3 Humoral Immunity

Studies of biomarkers of humoral immunity in children have consistently found significant associations between increasing blood lead concentration and serum immunoglobulin levels, with increasing serum IgE in association with increasing blood lead concentration (Table 6-8.1; Karmaus et al., 2005; Lutz et al., 1999; Sun et al., 2003). These effects were evident at blood lead concentrations <10 μg/dL. Increasing serum IgE levels also have been observed with increasing blood lead concentration (blood lead ≥30 μg/dL) in association with

Table 6-8.1. Summary of Results of Selected Studies of Associations Between Lead Exposure and Serum Immunoglobulin Levels

Study	Subjects	n[a]	Blood Lead (µg/dL)		IgA	IgE	IgG	IgM
			Mean (SD)	Range				
Children								
Annesi-Maesano et al. (2003)	neonates	374	67 (48)[b]	NR	NR	+[c]	NR	NR
Karmaus et al. (2005)	children, 7–10 yr	331	3	1–5[e]	o	+	o	o
Lutz et al. (1999)	children, 9 mo–6 yr	270	NR	1–45	NR	+	NR	NR
Sarasua et al. (2000)	children, 6–35 mo	372	7	~2–16[d]	+	NR	+	+
Sun et al. (2003)	children, 3–6 yr	73	NR	~3–40	NR	+	–	–
Adults								
Heo et al. (2004)	batter manufacture workers	606	~22 (~10)[e]	NR	o	+	o	o
Pinkerton et al. (1998)	smelter workers	229	39[f]	<2–55	o	NR	–	o
Sarasua et al. (2000)	general population	433	4.3	~1–10[d]	o	NR	o	o

-, decrease; +, increase; o, no effect; NR, not reported, Ig, serum immunoglobulin level

[a] total number of subjects (including reference group)
[b] infants cord blood (maternal blood lead mean was 96 µg/dL (SD 58)
[c] in association with increasing neonatal hair lead
[d] 5–95th percentile range
[e] mean of age-group means and SDs
[f] median

occupational exposures to lead (Heo et al., 2004). Outcomes for other immunoglobulin indices in adults have been less consistent (Pinkerton et al., 1998; Sarasua et al., 2000).

Possible associations between lead exposure and biomarkers of humoral immunity in children have been examined in several cross-sectional studies (Annesi-Maesano et al., 2003; Karmaus et al., 2005; Lutz et al., 1999; Reigart and Graher, 1976; Sarasua et al., 2000; Sun et al., 2003; Wagnerova et al., 1986). Four studies warrant particular attention because they examined a relatively low range of blood lead concentrations and applied multivariate analyses to the data in attempts to control for possible covariables (Karmaus et al., 2005; Lutz et al., 1999; Sarasua et al., 2000; Sun et al., 2003). Three studies found significant associations between increasing blood lead concentration and serum IgE levels (Karmaus et al., 2005; Lutz et al., 1999; Sun et al., 2003). The reported percent increase in serum IgE levels measured in these studies ranged from approximately 50 to 400% The Lutz et al. (1999) study measured serum IgE and IgG (against Rubella) in 270 children (age range 9 months to 2 years; blood lead range 1-45 μg/dL). The observed blood lead-age-IgE relationship is shown in Figure 6-8.1. The highest IgE levels (mean 211 IU/mL, SD 441, n = 17) were observed in children who had blood lead concentrations in the range 15–19 μg/dL; by comparison, mean IgE levels were blood lead concentrations in the range of 15–19 μg/dL; by comparison, mean IgE levels were 52 IU/mL (SD 166) for subjects who had blood lead concentrations <10 μg/dL (n = 174). The Karmaus et al. (2005) study measured serum IgA, IgE, IgG, and IgM levels in 331 children (age range 7-10 years). Blood lead concentrations were lower in this study than in the Lutz et al. (1999) study (1–5 μg/dL). A multivariate linear regression analysis revealed a significant association between blood lead ($p < 0.05$) and serum IgE, however, the change in serum IgE level was not monotonic with increasing blood lead concentration (Figure 6-8.2). The highest IgE levels (adjusted mean 59 IU/L) were observed in the children who had blood lead concentrations ranging from 2.8–3.4 μg/dL (n = 86) and >3.4 μg/dL (n = 82). Sun et al. (2003) measured serum IgE, IgG, and IgM levels in children, ages 3–6 years (blood lead concentration range 2.6–44 μg/dL, n = 73). A nonparametric comparison of immunoglobulin levels between low (<10 μg/dL) and high (≥10 μg/dL) blood lead strata revealed significantly higher IgE levels (Figure 6-8.3) and significantly lower IgG and IgM levels in the high blood lead stratum.

The study by Annesi-Maesano et al. (2003) provides further suggestive evidence for an association between lead exposure and increasing IgE levels. The study included 374

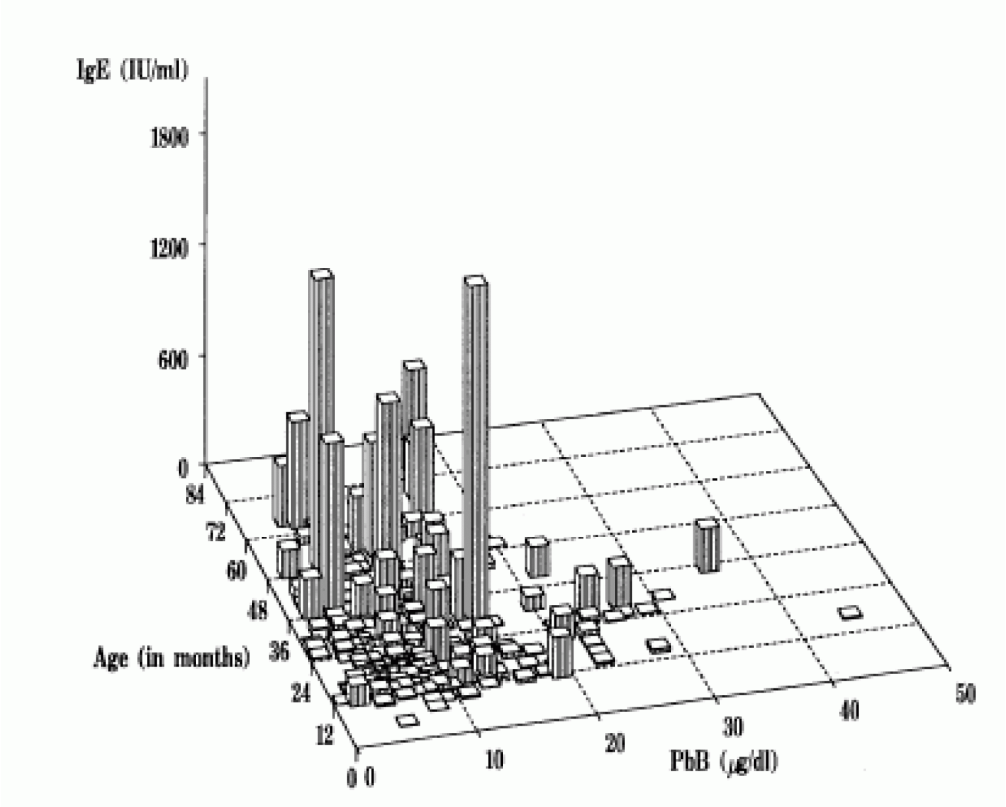

Figure 6-8.1. Relationship between blood lead concentration (PbB), age, and serum IgE level in children. Spearmen partial correlation between blood lead and serum IgE is 0.22 p = 0.0004, n = 221).

Source: Lutz et al. (1999).

mother-infant pairs who had relatively high mean blood lead levels (maternal mean 96 μg/dL, SD 58; infant cord 67 μg/dL, SD 48). Serum IgE level was significantly associated with increasing infant hair lead ($p < 0.001$), but not with cord blood lead or placental lead level. The association between IgE and hair lead levels was evident in a subset of mother-infant pairs, in which mothers were classified as nonallergenic, and was unrelated to maternal smoking (i.e., urinary cotinine).

The ATSDR Multisite Lead and Cadmium Exposure Study (ATSDR, 1995) is one of the largest studies to assess humoral immune status in association with lead exposures; however, it did not include an assessment of IgE. The study included a cross-sectional analysis of serum IgA, IgG, and IgM levels in 1,561 subjects (age range 6 months to 75 years) who resided in areas

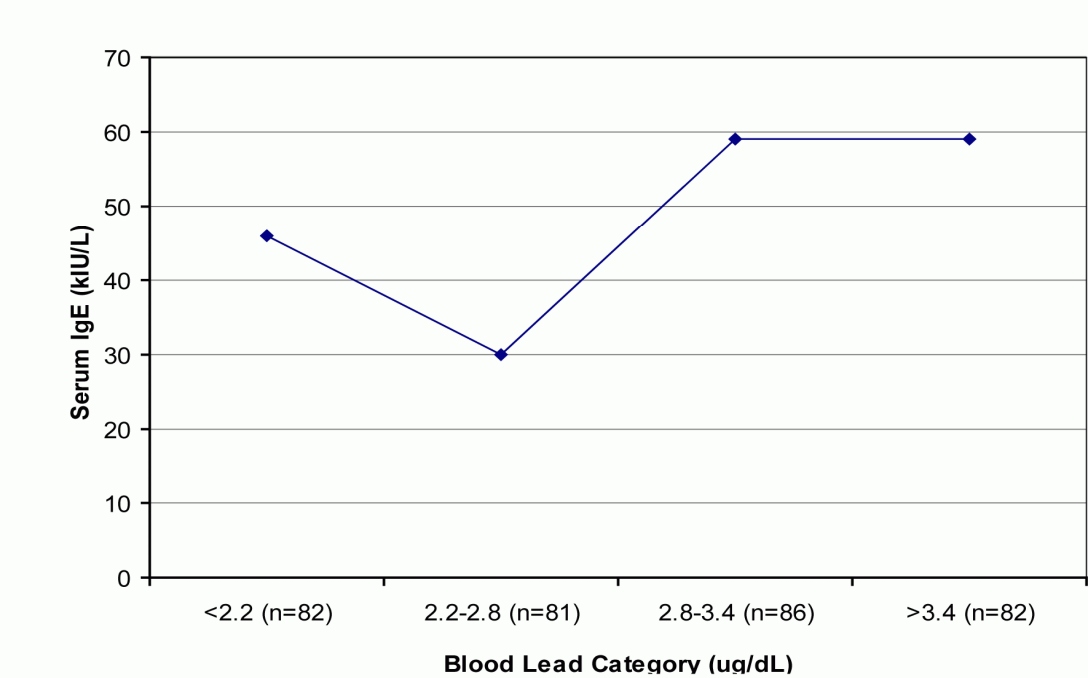

Figure 6-8.2. Relationship between blood lead concentration and serum IgE level in children. Mean serum IgE levels (standard deviations not reported) are adjusted for age, number of infections in the previous 12 months, exposure to passive smoke in the previous 12 months, and serum lipids (sum of cholesterol and triglycerides). Means of serum IgE levels in blood lead categories were significantly different (F-test p = 0.03).

Source: Karmaus et al. (2005).

impacted by lead mining and/or smelting operations and in 480 demographically-matched controls (Sarasua et al., 2000). A multivariate linear regression analysis of immunoglobulin levels and blood lead concentration (exposed and control groups combined) revealed associations between increasing blood lead and increasing serum IgA, IgG, and IgM levels in subjects 6–35 months of age (blood lead 5th–95th percentile range 1.7–16 µg/dL, Figure 6-8.4).

Possible associations between lead exposure and biomarkers of humoral immunity also have been examined in several cross-sectional studies of lead workers (Alomran and Shleamoon, 1988; Anetor and Adeniyi, 1998; Ayatollahi, 2002; Coscia et al., 1987; Ewers et al., 1982; Heo et al., 2004; Kimber et al., 1986; Pinkerton et al., 1998; Ündeger et al., 1996). Outcomes from

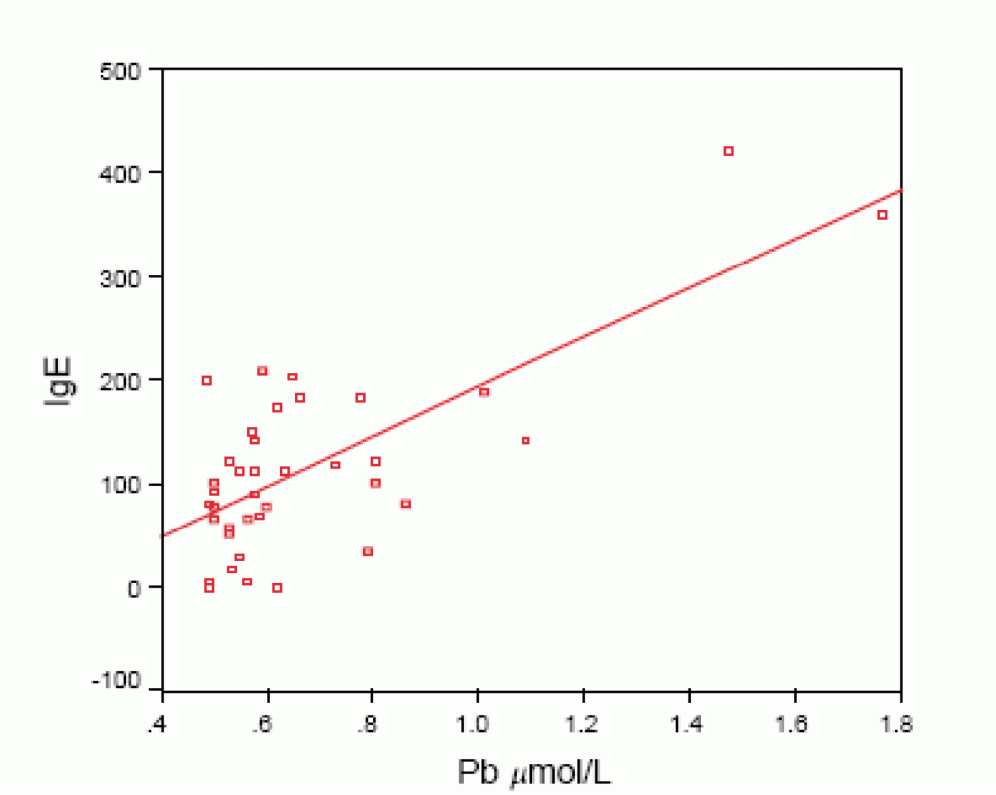

Figure 6-8.3. Relationship between blood lead concentration (lead) and serum IgE level in lead children. Mean serum IgE levels in female children whose blood lead concentrations were in the range 10–40 µg/dL (20.4 IU/L; n = 16) were significantly higher than for children whose blood lead concentrations <10 µg/dL (13.1 IU/L; n = 17).

Source: Sun et al. (2003).

these studies, with respect to humoral immune parameters, measured as serum and/or salivary immunoglobulin levels, are mixed. Some studies finding positive associations with blood lead (Heo et al., 2004), negative associations (Anetor and Adeniyi, 1998; Ewers et al., 1982; Pinkerton et al., 1998), or no (or mixed) effects (Alomran and Shleamoon, 1988; Kimber et al., 1986; Queiroz et al., 1994b; Sarasua et al., 2000; Ündeger et al., 1996).

Based on study design considerations (e.g., cohort criteria, size, treatments of covariates), three studies warrant particular attention (Heo et al., 2004; Pinkerton et al., 1998; Sarasua et al., 2000). Of these, only Heo et al. (2004) assessed serum IgE levels consistent with outcomes reported in children, increasing blood lead concentration was significantly associated with

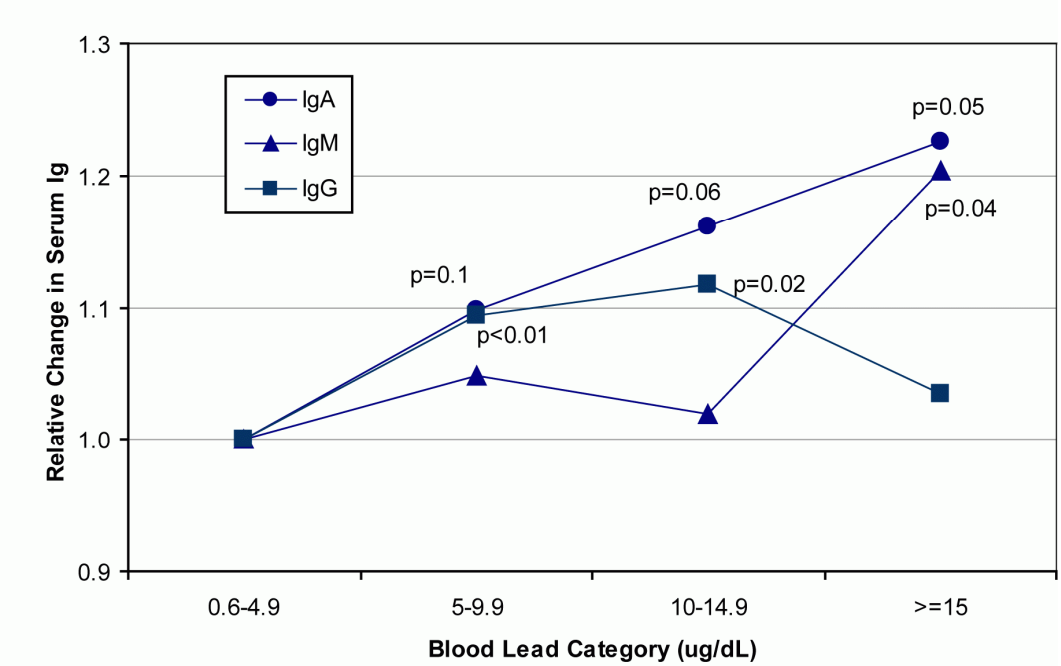

Figure 6-8.4. Relationship between blood lead concentration and serum immunoglobulin (Ig) levels in children. Shown are relative changes in serum Ig levels, adjusted for age, sex, and exposure location. P-values reflect comparison to <5 µg/dL blood lead category mean (<5 mg/dL, n = 165; 5-9.9 µg/dL, n = 136; 10–14.9 µg/dL, n = 47; ≥15 µg/dL, n = 24).

Source: Sarasua et al. (2000).

increasing serum IgE levels (Figure 6-8.5). The study measured serum IgE, IL-4 and IFNγ in 606 battery manufacture workers. Serum IgE levels were significantly higher in the blood lead stratum (≥30 µg/dL) compared to lower strata (<10 or 10–29 µg/dL) for the age strata 30–39 years, ≥40 years, and for all ages combined.

Although the Pinkerton et al. (1998) study did not assess IgE outcomes, it offers the strongest study design of the three for assessment of other immunoglobulin classes. Although it is a relatively small cross-sectional study, it considered immune illnesses and immune suppressant drugs in the construction of the cohorts and examined a relatively large number of potential covariates in the data analysis. Serum immunoglobulin levels were measured in male smelter (n = 145) workers and hardware workers (n = 84). Excluded (by blind evaluation) from the study cohorts were individuals who had "serious" illnesses of the immune system, who were

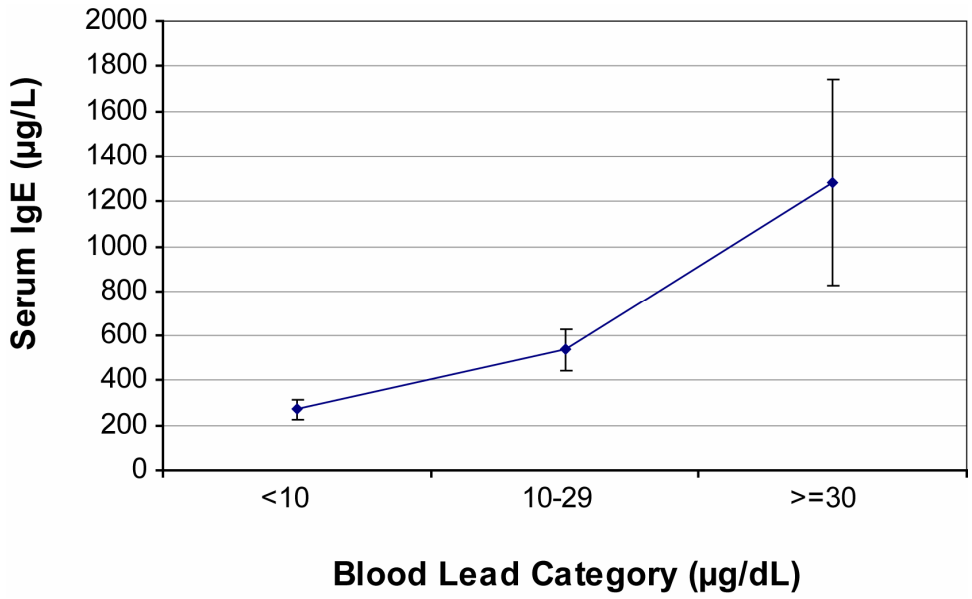

Figure 6-8.5. Relationship between blood lead concentration and serum IgE level in lead workers. Mean serum IgE levels in high blood lead category were significantly higher for all ages (shown), and within age categories ≥40 years and 30-39 years, but not within age category <30 years.

Source: Heo et al. (2004).

taking immune suppressant drugs, or who had chemical exposures (other than to lead) that might affect immune function. Median blood lead concentrations were 39 µg/dL (range 15–55) in the lead workers and <2 µg/dL (range <2–12) in the reference group. Covariate-adjusted (logistic regression) geometric mean serum IgA, IgG, and IgM, and salivary IgA levels in the lead workers were not significantly different from the reference group; however, the adjusted regression coefficient for serum IgG and time-integrated (but not current) blood lead concentration was negative and significant.

The Sarasua et al. (2000) study, described above for its assessment of children, also included a cross-sectional analysis of serum IgA, IgG, and IgM levels in adults (age 16–75 years, n = 433; blood lead 5th–9th percentile range 1–10 µg/dL) and found no significant associations between blood lead and serum immunoglobulin levels (serum IgE outcomes were not assessed).

Also germane to the evidence for effects of lead on humoral immunity in humans are the results of a clinical study in which serum immunoglobulin levels were repeatedly measured in a lead smelter worker who underwent CaEDTA therapy three times per week for a period of

10 weeks (Sata et al., 1998). Serum IgA, IgG, and IgM were significantly higher when assessed 24 h after each CaEDTA treatment compared to assessments made prior to treatment. Furthermore, serum IgG levels were significantly negatively correlated with blood lead concentration during the treatment period. Before-treatment and after-treatment blood lead concentration means were 45.1 (SD 16.0) and 31.0 (SD 9.8), respectively.

6.8.4 Cell-mediated Immunity

Studies of biomarkers of cellular immunity in children have found significant associations between increasing blood lead concentration and decreases in T-cell abundance, with corresponding increases in B-cell abundance (Karmaus et al., 2005; Sarasua et al., 2000; Zhao et al., 2004). These effects have been observed in children whose blood lead concentrations were below 10 µg/dL (Karmaus et al., 2005; Sarasua et al., 2000), although not all studies (e.g., Lutz et al., 1999) have found such associations at higher blood lead concentrations (e.g., 10–45 µg/dL). Studies of occupational lead exposures have also found associations between increasing blood lead concentration and changes in T-cell abundance (Fischbein et al., 1993; Pinkerton et al., 1998; Sata et al., 1997). Effects were observed in association with blood lead concentrations below 25 µg/dL (Fischbein et al., 1993) and in populations whose blood lead concentrations ranged from approximately 7 to 55 µg/dL (Pinkerton et al., 1998; Sata et al., 1997). Outcomes from these studies are qualitatively summarized in Table 6-8.2 and are discussed in greater detail below.

Several cross-sectional studies have examined possible associations between lead exposure and biomarkers of cellular immunity in children (Karmaus et al., 2005; Lutz et al., 1999; Sarasua et al., 2000; Zhao et al., 2004). Three studies (Karmaus et al., 2005; Sarasua et al., 2000; Zhao et al., 2004) found significant associations between increasing lead exposure and decreases in T-cell abundance (Table 6-8.2). The largest study (Sarasua et al., 2000) examined abundance of total lymphocytes, T-cells ($CD3^+$), B-cells ($CD20^+$), NK cells, and $CD4^+$ and $CD8^+$ T-cell phenotypes in infants, children, and adolescents. Associations between increasing blood lead concentration and increasing B-cell abundance (% and number), and decreasing T-cell abundance (%) were found for children 6–35 months of age (n = 312), after adjustment for age, sex, and study site (of four mining/smelting sites). Comparison of adjusted

Table 6-8.2. Summary of Results of Selected Studies of Associations Between Lead Exposure and Serum Lymphocyte Abundances

Study	Subjects	n[a]	Blood Lead (µg/dL)		T[b]	T_H[c]	T_C[d]	T_{HC}[e]	T_M[f]	NK[g]	B[h]
			Mean (SD)	Range							
Children											
Karmaus et al. (2005)	children, 7–10 yr	331	3	1–5[i]	–	o	–	NR	o	o	–
Lutz et al. (1999)	children, 9 mo–6 yr	270	NR	1–45	o	NR	NR	NR	NR	NR	o
Sarasua et al. (2000)	children, 6–35 mo	372	7	~2–16[i]	–	o	o	NR	NR	o	+
Zhao et al. (2004)	children, 3–6 yr	73	NR	~3–40	o	–	+	–	NR	NR	o
Adults											
Fischbein et al. 1993)	firearms instructors	87	31 (4)[j]	NR	–	–	o	NR	NR	o	+
Pinkerton et al. (1998)	smelter workers	229	39[k]	<2–55	o	o	o	o	+	o	+
Sarasua et al. (2000)	general population	433	4.3	~1–10[i]	o	o	o	o	NR	o	o
Sata et al. (1997)	lead stearate workers	99	19	7–50	o	o	+	NR	–	NR	o

–, decrease; +, increase; o, no effect; NR, not reported; SD, standard deviation.

[a] total number of subjects (including reference group)
[b] T-cells ($CD3^+$)
[c] T-helper cells ($CD4^+$)
[d] Cytotoxic T-cells ($CD8^+$)
[e] $CD4^+CD8^+$
[f] T-memory cells ($CD45RO^+$, $CD45RA^+$)
[g] Natural killer cells (e.g., $CD16^+$, $CD56^+$)
[h] B-cells (e.g., $CD19^+$, $CD20^+$)
[i] 5–95th percentile range
[j] high exposure group
[k] median

means for outcomes across blood lead strata revealed that the differences were significant for the ≥15 μg/dL stratum only, compared to the <5 μg/dL stratum. The Karmaus et al. (2005) study examined children in the age range 7–10 years (n = 331) who had blood lead concentrations <5 μg/dL. In addition to age and sex, regression models relating outcomes to blood lead concentration included exposure to environmental tobacco smoke and infections in the previous year as covariates. Similar to the Sarasua et al. (2000) study, Karmaus et al. (2005) found significant associations between blood lead concentration and decreased T-cell abundance ($CD3^+$, $CD3^+CD8^+$) and increased B-cell ($CD19^+$) abundance (for the blood lead quartile 2.2-2.8 μg/dL; Figure 6-8.6). Zhao et al. (2004) examined lymphocyte phenotype abundance in children in the age range 3–6 years (n = 73) and found significantly lower % abundance of T-cell phenotypes $CD3^+CD4^+$, $CD4^+CD8^+$ and significantly higher abundance of $D3^+CD8^+$ cells in children whose blood lead concentrations were ≥10 μg/dL compared to <10 μg/dL. Lutz et al. (1999) found no significant associations between blood lead concentration and age-adjusted T-cell ($CD3^+$) or B-cell (CD19+) abundance or abundance of various other lymphocyte phenotypes (i.e., $CD2^+$, $CD25^+$, $CD28^+$, $CD71^+$) in children whose blood lead concentrations were 10–14, 15–19, or 20–45 μg/dL compared to <10 μg/dL.

A larger set of studies have evaluated potential associations between lead exposure and biomarkers of cellular immunity in adults (Basaran and Ündeger, 2000; Cohen et al., 1989; Coscia et al., 1987; Fischbein et al., 1993; Kuo et al., 2001; Mishra et al., 2003; Pinkerton et al., 1998; Sarasua et al., 2000; Sata et al., 1998, 1997; Yücesoy et al., 1997b; Ündeger et al., 1996). Four studies warrant particular attention because they implemented relatively stronger study designs (i.e., cohort criteria, size, treatment of covariates): Fischbein et al., 1993; Pinkerton et al., 1998; Sarasua et al., 2000; and Sata et al., 1998). With one exception (Sarasua et al., 2000), all were studies of relatively small occupational cohorts. The Sarasua et al. (2000) study included a cross-sectional analysis of abundance of total lymphocytes, B-cells, NK cells, and $CD4^+$ and $CD8^+$ T-cell phenotypes in individuals (n = 433), age 16–75 years. Associations were not found between blood lead concentration and either B-cell or T-cell abundance, after adjustment for age, sex, and study site (of four mining/smelting sites). The study did detect significant associations among these variables in infants and children (see above discussion of cellular immunity outcomes in children). However, all three occupational studies found

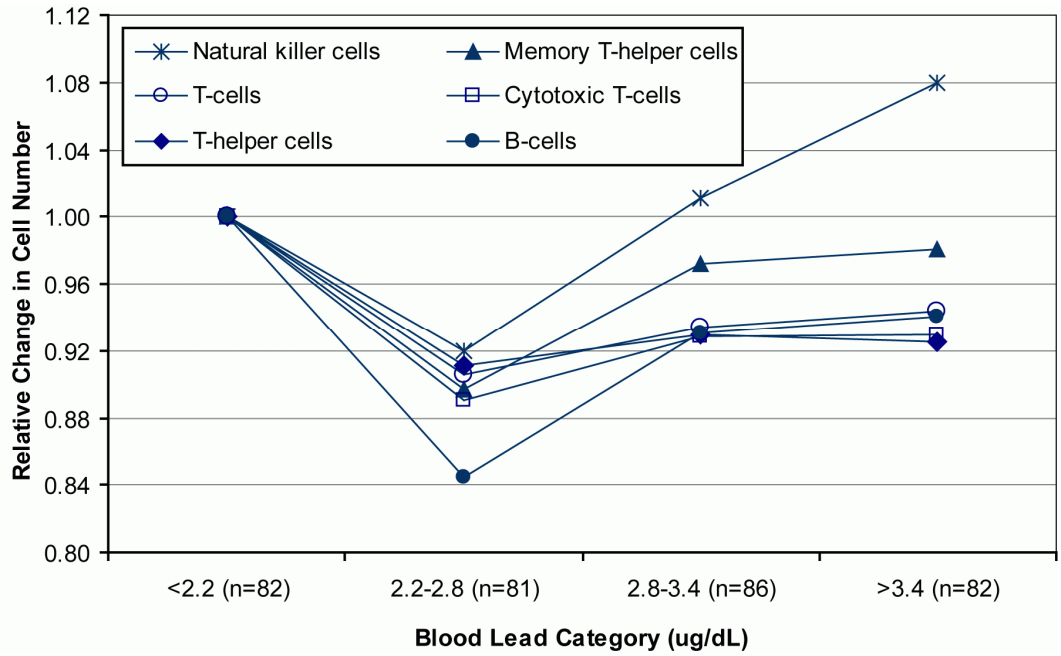

Figure 6-8.6. Relationship between blood lead concentration and T- and B-cell abundances in children. Shown are relative changes in covariate-adjusted absolute cell numbers (cells/µL) compared to the lowest blood lead group; adjusted for age, number of infections in the previous 12 months, exposure to passive smoke in the previous 12 months, and serum lipids (sum of cholesterol and triglycerides). Abundances for T-cells, cytotoxic T-cells, and B-cells in the 2.2-2.8 µg/dL group were significantly different ($p \leq 0.05$) from the <2.2 µg/dL group. Receptor phenotypes assayed were: T-cells, CD3+; T-helper cells, CD3+CD4+; cytotoxic T-cells, CD3+CD8+; memory T-helper cells, CD4+CD45RO+; natural killer cells, CD16+CD56+; B-cells, CD3+CD5+CD19+.

Source: Karmaus et al. (2005).

significant associations between increasing blood lead concentration and changes in abundanceof circulating T-cells with either no effect or an increasing B-cell abundance (Fischbein et al., 1993; Pinkerton et al., 1998; Sata et al., 1997). The strengths of the Pinkerton et al. (1998) study have been described previously with respect to outcome measures for humoral immunity. The study included male smelter workers (n = 145, mean blood lead 39 µg/dL; range 15–55) and hardware workers (n = 84, mean <2 µg/dL, range <2–12). Covariate-adjusted significant outcomes were an increase in B-cell ($CD19^+$) abundance (% and number) and increases in $CD4^+CD45RA^+$ cell

abundance (%, number) in association with increasing blood lead concentration. Covariate-adjusted mean levels of monocytes (%), and T-cells (% $D4^+CD8^+$, $CD8^+CD56^+$) were lower in lead workers compared to the reference group.

The Fischbein et al. (1993) study examined a small group of firearms instructors (n = 51) and age-matched reference subjects (n = 36). Fifteen of the instructors had blood lead concentration ≥25 µg/dL (mean 31.4, SD 4.3), the mean of the remaining 21 subjects was 4.6 µg/dL (SD 4.6). Mean blood lead concentration of the reference group was reported as <10 µg/dL. Increasing blood lead concentration was significantly associated with decreasing covariate-adjusted T-cell ($CD4^+$) abundance (Figure 6-8.7). Covariate-adjusted T-cell ($CD3^+$ % and number, $CD4^+$ % and number, $CD4^+CD8^+$ number) abundance was significantly lower and B-cell ($CD20^+$ cells % and number) abundance was higher in the instructors than in the reference group.

The Sata et al. (1998) study included male lead stearate manufacture workers (n = 71) and a nonexposed reference group (n = 28). Mean blood lead concentration was 19 µg/dL (range 7-50) in the lead workers (blood lead concentration for the reference group was not reported). Categorical covariate-adjusted lead exposure classification (exposed, not exposed) was significantly associated with lower T-cell ($CD3^+CD45RO^+$) number. Lead workers, relative to the reference group, had significantly lower covariate-adjusted mean $CD3^+CD45RO^+$ number and higher $CD8^+$ cells (%).

The above observations of decreasing T-cell abundance in association with lead exposure, as assessed from blood lead concentrations, is supported by results of several smaller cross-sectional studies, including Basaran and Ündeger (2000), Coscia et al. (1987), and Ündeger et al. (1996), as well as a clinical study in which T-cell and NK cell abundance was found to increase after CaEDTA chelation therapy of a lead smelter worker (Sata et al., 1997). Lower serum levels of the cytokines that function in the regulation of cellular immune responses, including IL-1β and IFN-γ, in lead workers compared to nonexposed subjects have also been observed (Yücesoy et al., 1997a).

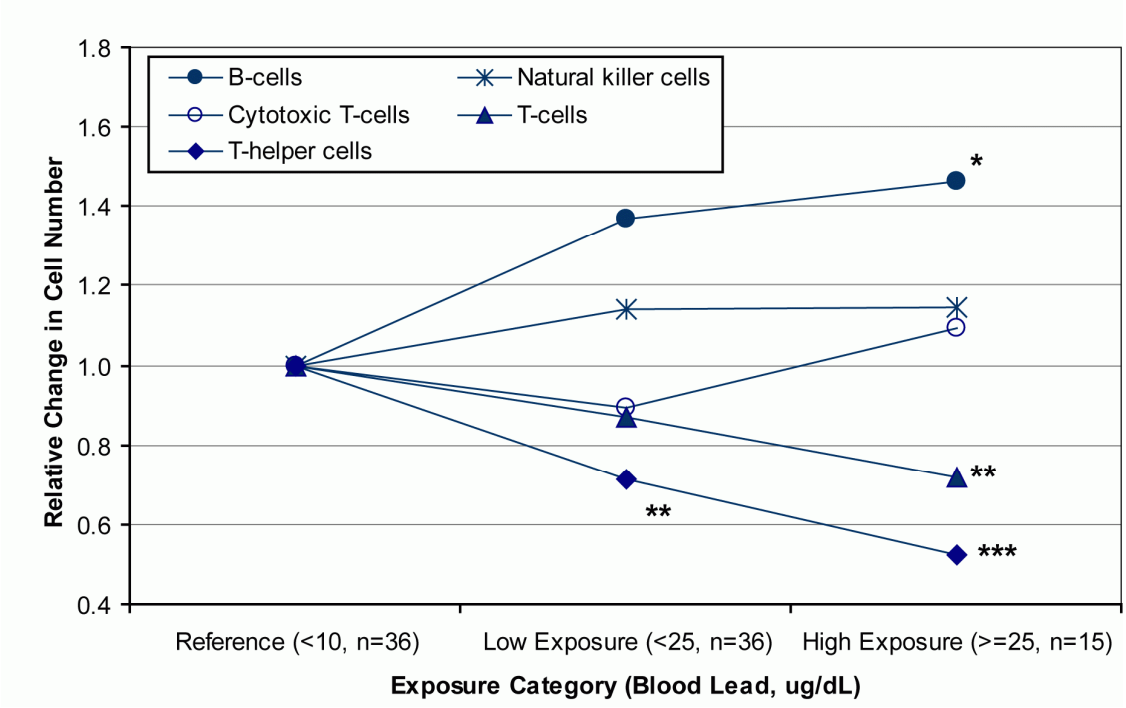

Figure 6-8.7. Relationship between lead exposure and T- and B-cell abundances in firearms instructors. Shown are relative changes in absolute cell numbers compared to the reference group. Comparisons of exposed relative to the reference group are shown as: * for $p < 0.05$; ** for $p < 0.01$; and * for $p < 0.002$. Receptor phenotypes assayed were: T-cells, CD3+; T-helper cells, CD4+; cytotoxic T-cells, CD8+; natural killer cells, CD16+; B-cells, CD20+. The CD4+/CD8+ ratio (not shown) was significantly lower in both the low exposure (1.38 [SD 0.5], $p < 0.002$) and higher exposure group (0.95 [SD 0.5], $p < 0.002$), compared to the reference group (1.95 [SD 0.66]).**

Source: Fischbein et al. (1993).

6.8.5 Lymphocyte Function

Limited evidence from occupational studies suggests that lead may suppress lymphocyte activation in humans. All available studies are of adults. Several studies have examined associations between lead exposure in adults and lymphocyte activation, assessed as a proliferative response to mitogens and/or antigens (Alomran and Shleamoon, 1988; Cohen et al., 1989; Fischbein et al., 1993; Kimber et al., 1986; Mishra et al., 2003; Pinkerton et al., 1998; Queiroz et al., 1994b). Results of these have been mixed. Three studies found no significant associations between blood lead concentrations in lead workers and lymphocyte proliferative

response to activating agents (; Kimber et al., 1986; Pinkerton et al., 1998; Queiroz et al., 1994b). Four studies found decreasing proliferative response with increasing blood lead concentration (Alomran and Shleamoon, 1988; Cohen et al., 1989; Fischbein et al., 1993; Mishra et al., 2003). The Alomran and Shleamoon (1988), Cohen et al. (1989), Mishra et al. (2003), and Queiroz et al. (1994b) studies, which found significant lead associations, included subjects who had relatively high blood lead levels (>60 µg/dL) compared to the Kimber et al. (1986) and Pinkerton et al. (1998) studies. The inclusion of subjects with higher lead concentrations may have contributed to the differences in outcomes.

As noted in the previous section, the Fischbein et al. (1993) and Pinkerton et al. (1998) studies are particularly noteworthy because of the strengths of the cohort selection and the data analyses which attempted to account for potential confounders. Also, these are the only reported studies that examined antigen-specific lymphocyte activation in humans. Mean blood lead concentrations in the two studies were similar 31 µg/dL (SD 4) in the Fischbein et al. (1993) study and 39 µg/dL (range 15–55) in the Pinkerton et al. (1998) study. Both studies found no significant associations between lead exposures (i.e., blood lead concentration) and antigen-specific lymphocyte proliferation, assessed in the Pinkerton et al. (1998) study with tetanus toxoid as the antigen and in the Fischbein et al. (1993) study with staphylococcus aureus as the antigen. However, the Fischbein et al. (1993) study also measured mitogen-induced lymphocyte proliferation (induced with PHA or PWM) and found a significantly lower proliferative response to the mitogens in association with lead exposure. This study also found a significant association between increasing blood lead concentration and decreasing proliferative response in mixed lymphocyte cultures (i.e., proliferative response of lymphocytes from exposed subjects when incubated with inactivated lymphocytes from a reference subject).

Inorganic lead has been shown by in vitro studies to perturb several aspects of lymphocyte function when introduced into primary isolates of human blood monocytes. Activated lymphocytes show altered lysosomal enzyme secretion and altered expression and secretion of cytokines (Bairati et al., 1997; Guo et al., 1996a; Hemdan et al., 2005). Lymphocytes activated with *Salmonella enteritidis* or to monoclonal antibodies of CD3, CD28 and CD40, and exposed to inorganic lead had suppressed expression of T-helper cell type T_H-1 cytokines, interferon (IFN-γ), interleukin (IL-1β), and tumor necrosis factor (TNF-α), whereas activation by CD antibodies increased secretion of T_H-2 cytokines, IL-5, IL-6, and IL-10 (Hemdan et al. 2005).

Inorganic lead also activates transcription factor NK-κβ in $CD4^+$ cells (Pyatt et al., 1996), an important regulator of T-cell activation, and increases expression of MHC class II surface antigens (HLA-DR), an important surface antigen in the $CD4^+$ response to exogenous antigens (Guo et al., 1996b). Lead increases antibody production in cultured human B-cells (McCabe and Lawrence, 1991). These observations suggest that they may perturb cellular immune function through a variety of mechanisms.

6.8.6 Phagocyte (Macrophage and Neutrophil) Function

Studies of lymphocyte and phagocyte (i.e., macrophage, neutrophil) function have found associations between blood lead concentrations and suppressed activation of macrophages in children whose blood lead concentrations ranged from 4 to 50 μg/dL (Pineda-Zavaleta et al., 2004). In addition, studies have observed suppressed PMNL chemotaxis in association with occupational exposures that resulted in blood lead concentrations of 12–90 μg/dL (Bergeret et al., 1990; Queiroz et al., 1994a, 1993).

Pineda-Zavaleta et al. (2004) examined mitogen (PHA)- and cytokine (INFγ)-induced activation of blood monocytes collected from 65 children (age range 6–11 years) who resided near an active lead smelter. Mean blood lead concentrations of subjects at three schools were: 7.0 μg/dL (range 3–25 μg/dL; 8,100 meters from smelter complex), 21 μg/dL (range 11-49 μg/dL; 1,750 meters from smelter), and 30 (range 10–48 μg/dL; 650 meters from smelter). Endpoints measured were nitric oxide and superoxide anion production, a response generally attributed to activated macrophages. Increasing blood lead concentration was significantly associated with decreasing PHA-induced nitric oxide production and increasing INFγ-induced superoxide anion production. The mitogen, PHA, activates macrophages indirectly through activation of lymphocytes, whereas INFγ, a cytokine released from CD44 (T_H1) cells, directly activates macrophages. Thus, one interpretation of this outcome is that lead suppressed T-cell mediated macrophage activation and stimulated cytokine-induced macrophage activation.

Possible associations between occupational lead exposure and PMNL chemotaxis and phagocytic activity have been explored in several small cross-sectional studies. Consistent findings are significantly reduced chemotactic response and phagocytic activity (i.e., respiratory burst, luminal uptake) in lead workers compared to reference groups. The largest study is that of Queiroz et al. (1994a, 1993) which evaluated PMNL function in several (possibly overlapping)

cohorts of lead battery manufacture workers (n = 60). Blood lead concentrations in the study groups ranged from 12 to 90 µg/dL. PMNL chemotaxis and lytic activity were significantly lower in the lead workers compared to the reference group. Bergeret et al. (1990) assessed PMNL chemotaxis and phagocytosis in a group of battery smelting workers (n = 34) and in a group of reference subjects (n = 34) matched to the lead worker group by age, sex, ethnic origin, smoking and alcohol consumption habits, and intake of antibiotics and NSAIDs. Mean blood lead concentrations were 71 µg/dL (SD 18) in the lead workers and 9 µg/dL (SD 4) in the reference group. Significantly lower PMNL chemotactic response to FMLP and phagocytic response in opsonized zymosan were significantly lower in the lead workers than in the reference group. Lead introduced into primary cultures of human PMNLs suppressed chemotaxis and phagocytosis (Governa et al., 1987).

6.8.7 Summary of the Epidemiologic Evidence for the Effects of Lead on the Immune System

Several studies have examined possible associations between lead exposures and biomarkers of immune function. Findings from the epidemiologic studies suggest that lead exposure (as reflected in blood lead concentration) may be associated with effects on cellular and humoral immunity. These effects include changes in serum immunoglobulin levels; perturbation of peripheral lymphocyte phenotype profiles, including decreases in peripheral blood T-cell abundance and changes in T-cell:B-cell abundance ratios; suppression of lymphocyte activation; and suppression of neutrophil chemotaxis and phagocytosis.

Studies of biomarkers of humoral immunity in children have consistently found significant associations between increasing blood lead concentration and serum Ig levels with increasing serum IgE in association with increasing blood lead concentration (Karmaus et al., 2005; Lutz et al., 1999; Sun et al., 2003). These effects were evident at blood lead concentrations below 10 µg/dL. Findings of studies of adults have been mixed with significant associations between blood lead (>30 µg/dL) and serum immunoglobulin levels (Heo et al., 2004; Pinkerton et al., 1998) and no association in a study group in which blood lead concentrations were <10 µg/dL (Sarasua et al., 2000).

Studies of biomarkers of cellular immunity in children have found significant associations between increasing blood lead concentration and decreases in T-cell abundance, with

corresponding increases in B-cell abundance (Karmaus et al., 2005; Sarasua et al., 2000; Zhao et al., 2004). These effects have been observed in children whose blood lead concentrations were below 10 µg/dL (Karmaus et al., 2005; Sarasua et al., 2000), although not all studies have found such associations at higher blood lead concentrations (e.g., 10–45 µg/dL; Lutz et al., 1999). Studies of occupational lead exposures have also found associations between increasing blood lead concentration and decreasing T-cell abundance (Pinkerton et al., 1998; Sata et al., 1997; Fischbein et al., 1993). Effects were observed in association with blood lead concentrations below 25 µg/dL (Fischbein et al., 1993) and in populations whose blood lead concentrations ranged from approximately 7 to 55 µg/dL (Pinkerton et al., 1998; Sata et al., 1997).

Studies of lymphocyte and phagocyte (i.e., macrophage, neutrophil) function have found associations between blood lead concentrations and suppressed activation of macrophages in children whose blood lead concentrations ranged from 4 to 50 µg/dL (Pineda-Zavaleta et al., 2004); suppressed PMNL chemotaxis in association with occupational exposures that resulted in blood lead concentrations of 12 to 90 µg/dL (Bergeret et al., 1990; Queiroz et al., 1994a, 1993), and suppressed mitogen-induced activation of peripheral lymphocytes in adults in association with occupational exposures that resulted in blood lead concentrations that ranged from 15 to 55 µg/dL (Fischbein et al., 1993).

6.9 EFFECTS OF LEAD ON OTHER ORGAN SYSTEMS

6.9.1 Biochemical Effects of Lead

6.9.1.1 Summary of Key Findings of the Biochemical Effects of Lead from the 1986 Lead AQCD

The 1986 Lead AQCD provided an extensive discussion of the effects of lead on heme biosynthesis and on quantitative relationships between exposure and effects in humans. Lead interferes with heme synthesis by inhibiting the enzymes δ-aminolevulinic acid dehydratase (ALAD) and ferrochelatase. As a consequence, heme biosynthesis decreases, relieving the rate-limiting enzyme of the heme synthesis pathway, δ-aminolevulinic synthetase (ALAS), from negative feedback inhibition by heme (Figure 6-9.1). The outcomes of decreased activity of ALAD and ferrochelatase, and increased activity of ALAS are increased urinary excretion of

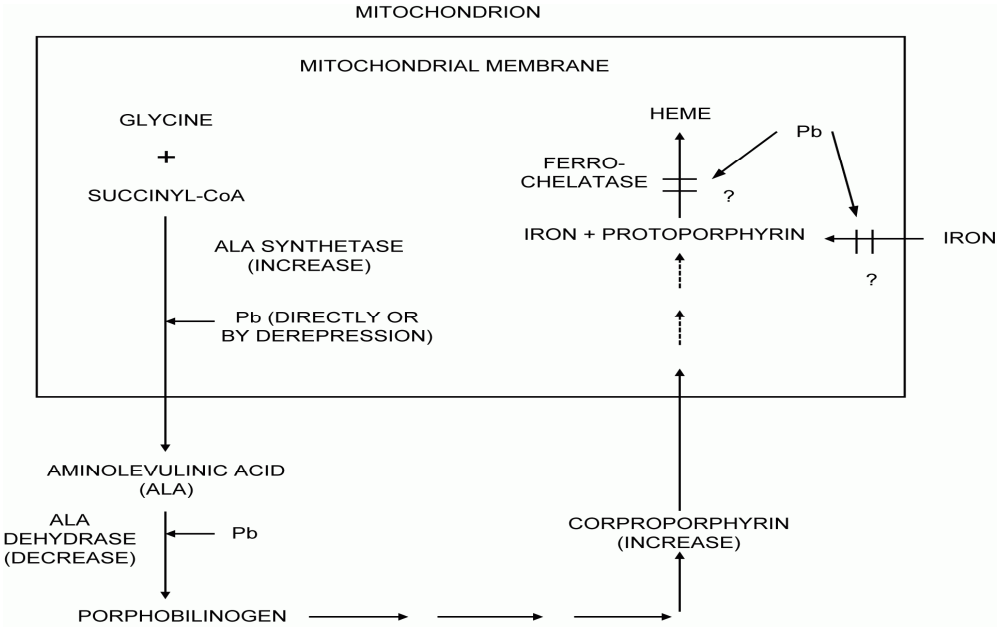

Figure 6-9.1. Effects of lead on heme biosynthesis.

Source: Derived from EPA (1986).

coproporphyrin (CP) and δ-aminolevulinic acid (ALA), increased level of ALA in blood plasma, and increased erythrocyte protoporphyrin (EP) levels.

Associations between lead exposure and blood ALAD activity and EP levels, and urinary ALA and CP excretion have been studied extensively in adults and children, and quantitative relationships between exposure and effect are well understood. Much of this information was available prior to completion of the 1986 Lead AQCD and is summarized in that criteria document (e.g., Alessio et al., 1976; Hernberg et al., 1970; Lilis et al., 1978; Piomelli et al., 1982; Roels et al., 1979; Selander and Cramer, 1970; Valentine et al., 1982). Numerous studies published since the 1986 AQCD provide additional support for the lead concentration-response relationships in humans described in the 1986 AQCD. The most pertinent studies are summarized in Annex Tables AX6-9.1 and AX6-9.2. The studies that provide the strongest basis for empirically-derived expressions relating blood lead concentration, blood ALAD activity, urinary ALA, and EP are listed in Table 6-9.1 and are discussed below.

Table 6-9.1. Blood Lead–Response Relationships for Heme Synthesis Biomarkers in Adults and Children

Study	n	Age	Blood Lead (µg/dL)	Regression Equation (r)	Blood Lead Change (µg/dL) Predicted to Halve or Double Effect Biomarker
ALAD Activity Decrease					
Roels and Lauwerys (1987)	143	10–13 yr	5–41	log[ALAD]= 1.864–0.015[blood lead] (r = 0.87)	20.1
Alessio et al. (1976, 1977)	169	Adult (m)	15–150	log[ALAD]=3.73–0.031[blood lead] (r = 0.87)	22.4
Hernberg et al. (1970)	158	Adult (m, f)	5–95	log[ALAD]=2.274–0.018[blood lead] (r = 0.90)	16.1
Morita et al. (1997)	58	Adult (m)	2–82	log[ALAD]=1.8535–0.00971[blood lead] (r = 0.76)	20.1
Urinary ALA Increase					
Roels and Lauwerys (1987)	37	10–13 yr	20–41	log[ALAU]=0.94+0.11[blood lead] (r = 0.54)	20.9
Alessio et al. (1976, 1977)	316	Adult (m)	10–150	log[ALAU]=1.25+0.014[blood lead] (r = 0.62)	49.5
Gennart et al. (1992)	183	Adult (m, f)	4–75	log[ALAU]=0.37+0.008[blood lead] (r = 0.64)	37.6
Oishi et al. (1996)	418	Adult (m, f)	10–99	log[ALAU]= -0.387+0.022[blood lead] (r = 0.71)	13.7
Selander and Cramer (1970)	150	Adult (m, f)	6–90	log[ALAU]= -1.0985+0.157[blood lead] (r = 0.74)	19.2
Roels and Lauwerys (1987)	39	Adult (m)	10–60	log[ALAU]=0.37+0.006[blood lead] (r = 0.41)	50.2
Roels and Lauwerys (1987)	36	Adult (f)	7–53	log[ALAU]=0.15+0.015[blood lead] (r = 0.72)	20.1

Table 6-9.1 (cont'd). Blood Lead–Response Relationships for Heme Synthesis Biomarkers in Adults and Children

Study	n	Age	Blood Lead (µg/dL)	Regression Equation (r)	Blood Lead Change (µg/dL) Predicted to Halve or Double Effect Biomarker
EP Increase					
Piomelli et al. (1982)	2,002	2–12	2–98	log[EP]=1.099+0.016[blood lead] (r = 0.509)	18.8
Roels and Lauwerys (1987)	51	10–13	15–41	log[EP]=1.321+0.025[blood lead] (r = 0.73)	12.0
Soldin et al. (2003)	4,908	0–17	<1–103	$EP = -0.0015[PbB]^3+0.1854[\text{blood lead}]^2-2.7554[PbB]+30.911$ (r = 0.999)	20.6
Alessio et al. (1976, 1977)	95	Adult (m)	10–90	log[EP]=0.94+0.0117[blood lead]	25.7
Alessio et al. (1976, 1977)	93	Adult (f)	10–70	log[EP]=1.60+0.0143[blood lead]	21.1
Gennart et al. (1992)	183	Adult (m)	4–75	log[EP]=0.06+0.019[blood lead] (r = 0.87)	15.8
Roels and Lauwerys (1987)	39	Adult (m)	10–60	log[EP]=1.41+0.014[blood lead] (r = 0.74)	21.1
Roels and Lauwerys (1987)	36	Adult (f)	7–53	log[EP]=1.23+0.027[blood lead] (r = 0.81)	11.1
Wildt et al. (1987)	851	Adult (m)	10–80	log[EP]=1.21+0.0148[blood lead] (r = 0.72)	20.3
Wildt et al. (1987)	139	Adult (f)	10–80	log[EP]=1.48+0.0113[blood lead] (r = 0.56)	20.6

ALA, δ-aminolevulinic acid; ALAD, δ-aminolevulinic acid dehydratase; ALAU, urinary δ-aminolevulinic acid; EP, erythrocyte protoporphyrin; PbB, blood lead concentration

Since completion of the 1986 Lead AQCD, a literature has developed on the effects of lead on serum and blood lipids, including cholesterol levels and indications of oxidative stress, in the form of lipid peroxides, depletion of erythrocyte reduced glutathione (GSH), and production of reactive oxygen species (ROS). These studies also are summarized in Annex Tables AX6-9.1 and AX6-9.2, and key findings are discussed below.

6.9.1.2 Heme Biosynthesis

6.9.1.2.1 ALAD Inhibition

Numerous studies published since completion of the 1986 AQCD have explored associations between lead exposure and inhibition of ALAD activity, as assessed from measurements of blood ALAD activity (Gurer-Orhan et al., 2004; Kim et al., 2002; Lee et al., 2000; Makino et al., 1997; Roels and Lauwerys, 1987; Schuhmacher et al., 1997), or urinary ALA excretion (Gennart et al., 1992; Oishi et al., 1996; Schuhmacher et al., 1997; Wildt et al., 1987; Soldin et al., 2003). Quantitative estimates derived from the larger, more recent studies are presented in Table 6-9.1. Blood lead concentration is inversely correlated with the log of blood ALAD activity and log of urinary ALA and quantitative estimates of the change in blood ALAD activity per unit change in blood lead concentration are consistent across studies. Halving of blood ALAD activity occurs with an increase in blood lead concentration of approximately 20 μg/dL in both children (Roels and Lauwerys, 1987) and adults (Morita et al., 1997). These estimates are consistent with earlier studies of adults (e.g., Hernberg et al., 1970) and children (e.g., Alessio et al., 1976, 1977), discussed in the 1986 AQCD. Greater variability is apparent in estimates of the change in urinary ALA per unit change in blood lead concentration (Table 6-9.1). This may be related, in part, to gender-heterogeneity in the relationship. Roels and Lauwerys (1987) estimated that urinary ALA doubles in association with a 20 μg/dL increase in blood lead concentration in females and 50 μg/dL in males. In a much larger study (Oishi et al., 1996), an analysis that combined data from males (n = 253) and females (n = 165) found that a doubling of urinary ALA occurred in association with a 14 μg/dL increase blood lead concentration. Urinary ALA excretion increases as a linear function of plasma ALA concentration (Oishi et al., 1996); thus, the gender heterogeneity for the blood lead-urinary ALA relationship may derive from a gender difference in the effect of lead on plasma ALA concentration or from differences in renal plasma clearance of ALA.

6.9.1.2.2 ALAD Polymorphism

ALAD is a polymorphic enzyme with two alleles (ALAD1 and ALAD2) and three genotypes: ALAD1,1, ALAD1,2, and ALAD2,2 (Battistuzzi et al., 1981). The corresponding phenotypes appear to have nearly identical catalytic properties (Battistuzzi et al., 1981). The predominant genotype is ALAD1,1 which has a prevalence of approximately 90% (Astrin et al., 1987; Battistuzzi et al., 1981; Hsieh et al., 2000; Shen et al., 2001). A higher percentage of erythrocyte lead was bound to ALAD in carriers of the ALAD2 allele (84%) compared to carriers of the ALAD1 allele (81%); however, no differences were evident in the distribution of lead between erythrocytes and plasma (Bergdahl et al., 1997), and there is no evidence that the ALAD genotype confers different sensitivity to inhibition of heme biosynthesis (Hsieh et al., 2000; Perez-Brava et al., 2004; Schwartz et al., 1997; Suzen et al., 2003).

6.9.1.2.3 Ferrochelatase Inhibition

Lead inhibition of ferrochelatase results in an accumulation of protoporphyrin IX in erythrocytes (EP, also referred to as zinc protoporphyrin, or ZPP, or iron protoporphyrin, FEP, depending on the method used to make the measurement). Numerous studies have examined relationships between blood lead concentration and EP levels in adults and children. Quantitative estimates based on the most pertinent studies are presented in Table 6-9.1. Results across these studies are similar. In both children and adults (males and females), a doubling of EP levels occurs in association with an increase in blood lead concentration of approximately 20 µg/dL (Piomelli et al., 1982; Soldin et al., 2003; Wildt et al., 1987). A pronounced gender difference in the relationship between EP and blood lead concentration was observed by Roels and Lauwerys (1987) which was not observed in the much larger study of Wildt et al. (1987).

Inhibition of ferrochelatase also gives rise to an increase in urinary coproporphyrin, with a similar relationship to blood lead concentration; a doubling of urinary EP occurs in association with an increase in urinary coproporphyrin of approximately 20 µg/dL (Alessio et al., 1976).

6.9.1.3 Effects on Blood Lipids

Associations between occupational exposure to lead and changes in blood lipid composition have been observed. These include increased levels of lipid peroxides in blood and/or serum (Ito et al., 1985; Jiun and Hsien, 1994; Sugawara et al., 1991) and increased serum

levels of total and HDL cholesterol (Kristal-Boneh et al., 1999). Increased levels of glucose-6-phosphate dehydrogenase (G6PD) in erythrocytes have also been observed in lead workers (Cocco et al., 1995; Gurer-Orhan et al., 2004).

Kristal-Boneh et al. (1999) measured serum total, HDL, and LDL cholesterol, and triglycerides in a group of male battery manufacture workers. Covariate-adjusted serum total-cholesterol and HDL cholesterol levels were 6% and 12% higher, respectively, in lead workers (n = 56, mean blood lead 42 μg/dL, SD 15) compared to reference group (mean blood lead: 2.7 μg/dL). Increasing blood lead concentration was significantly associated with increasing covariate-adjusted total cholesterol and HDL cholesterol. A similar outcome was found in a larger study (Ito et al., 1985) of male steel workers (n = 712, blood lead range 5–62 μg/dL). When stratified by age, total and HDL cholesterol levels in serum were 3.6% and 7.5% higher, respectively, in lead workers in the age range 40 to 49 years, compared to corresponding strata of the office workers (n = 155). Although a smaller study, the Kristal-Boneh et al. (1999) study considered a larger set of potential covariables (e.g., dietary fat, cholesterol, and calcium intakes, sport activities, alcohol consumption, cigarette smoking).

Oxidative changes in blood lipids (e.g., increased levels of lipid peroxides and malondialdehyde levels) as well as decreased levels of erythrocyte superoxide dismutase (SOD), catalase, G6PD, and GSH peroxidase, indicative of increased oxidative stress, have been observed in lead workers, in comparison to reference groups (Ito et al., 1985; Jiun and Hsien, 1994; Solliway et al., 1996; Sugawara et al., 1991). However, none of these studies have developed concentration-response relationships that take into account potential confounders. The largest study is that of (Ito et al., 1985), described above. When stratified by age, serum HDL cholesterol and serum lipoperoxide levels were 16% higher in the lead workers in the age range 40 to 49 years, compared to corresponding strata of the reference group. Serum lipoperoxide levels also appeared to increase as blood lead increased above 30 μg/dL, while erythrocyte SOD appeared to decrease with increasing blood lead concentration (a statistical evaluation was not reported).

Evidence for increased oxidative stress (increased reactive oxygen species) in lymphocytes of lead workers has also been reported (Fracasso et al., 2002). Peripheral lymphocytes collected from battery manufacture workers (n = 37, mean blood lead: 40 μg/dL) exhibited increased DNA strand breaks, higher production of ROS and lower GSH levels

compared to a reference group of office workers (n = 29, mean blood lead 4 μg/dL). The covariate-adjusted odds ratios (exposed versus not exposed) were 1.069 (95% CI: 1.020, 1.120) for increased DNA strand breaks and 0.634 (95% CI: 0.488, 0.824) for lower GSH levels.

6.9.2 Effects of Lead on the Hematopoietic System

6.9.2.1 Summary of Key Findings of the Effects of Lead on the Hematopoietic System from the 1986 Lead AQCD

The 1986 Lead AQCD concluded that lead decreases heme production and shortens erythrocyte survival; both effects contributing to lead-induced anemia in children and adults, which becomes evident in children at blood lead concentrations ≥40 μg/dL and, in adults, ≥50 μg/dL. The 1986 Lead AQCD also concluded that effects of lead on blood hemoglobin level extend below 50 μg/dL, with effects detected in lead workers at blood lead concentrations <25 μg/dL (Baker et al., 1979; Grandjean, 1979). More recent epidemiologic studies, summarized below, provide additional information on concentration-response relationships for hematopoietic effects of lead. The studies support the conclusion that clinical anemia can occur in children in association with blood lead concentrations >40 μg/dL (Schwartz et al., 1990). The newer studies suggest that perturbation of erythropoiesis, indicated by changes in serum erythropoietin, occurs in association with blood lead concentrations <40 μg/dL and in the absence of detectable changes in blood hemoglobin levels or hematocrit. Details regarding the design of these studies and outcomes are presented in Annex Tables AX6-9.3 and AX6-9.4. Outcomes of the most pertinent studies are discussed below.

6.9.2.2 Blood Hemoglobin Levels

Several studies reported since the completion of the 1986 Lead AQCD have explored associations between lead exposure and blood hemoglobin levels in children and adults. Consistent findings have been a lack of discernable depression of blood hemoglobin levels in study populations whose mean blood lead concentrations were ≤40 μg/dL (Table 6-9.2). Of note is the findings relating patella bone lead to both blood hemoglobin levels and hematocrit.

The Kosovo prospective study of pregnancy outcomes is one of the largest epidemiologic evaluations of associations between lead exposure and blood hemoglobin levels in infants and children (Graziano et al., 2004; Factor-Litvak et al., 1999, 1998). The study included pregnant

Table 6-9.2. Summary of Results of Selected Studies of Associations Between Lead Exposure and Blood Hemoglobin Levels

Study	Subjects	n[a]	Blood Lead (µg/dL)		Blood Hemoglobin	Comment
			Mean (SD)	Range		
Children						
Graziano et al. (2004)	ages: 4.5–12 yr	311	6–9, 31–39[b]	3–70	o	+ erythropoietin
Liebelt et al. (1999)	ages: 1–6 yr	86	18[c]	2–84	o	– erythropoietin
Adults						
Graziano et al. (1990)	pregnant women	1,502	5, 17[d]	2–43	o	– erythropoietin
Hu et al. (1994)	male carpenters	119	8	2–25	o	– in association with patella bone lead
Makino et al. (1997)	male VCS workers	1,573	13	1–39	+	(+) 1 g/dL per 10 µg/dL blood lead
Solliway et al. (1996)	male battery workers	100	10	23–63	o	– RBC count
Gennart et al. (1992)	battery workers	183	51 (8)	40–70	–	– hematocrit
Horiguchi et al. (1991)	male lead refinery workers	40	54 (16)	NR	–	– hematocrit
Poulos et al. (1986)	male lead workers	160	18–27 (5)[e]	NR	–	– hematocrit

-, decrease; +, increase; Hgb, hemoblogin; NR, not reported; PCV, packed cell volume SD, standard deviation; VCS, vinyl chloride stabilizer

[a] total number of subjects (including reference group)
[b] range of means of low and higher exposure groups
[c] median
[d] mean of low- and high-exposure groups
[e] range of group means (standard deviation estimated for up range based on reported standard error).

women (n = 1502) and their children (n = 311) who resided in one of two regions of Kosovo, Yugoslavia; one was heavily impacted by lead industries (high-lead area), the other had relatively little lead contamination (low-lead area). Mean blood lead concentrations of children (measured at birth and at intervals to 12 years of age) ranged from 30 to 40 μg/dL in the high-lead area and 6 to 9 μg/dL in the low-lead area. Mean blood hemoglobin levels in the low-lead and high-lead children, measured at 4.5, 6.5, 9.5, and 12 years of age, were not significantly different. These findings are consistent with those from a smaller cross-sectional study (n = 89; blood lead range 2 to 84 μg/dL, 84% <35 μg/dL) that also found no association between blood lead concentration and blood hemoglobin levels (Liebelt et al., 1999). Results from these two studies suggest that, in the absence of iron deficiency, lead exposures that result in blood lead concentrations <40 μg/dL do not produce detectable changes in blood hemoglobin levels in children.

Associations between lead exposure and blood hemoglobin levels in adults have been examined in numerous epidemiological studies (Froom et al., 1999; Gennart et al., 1992; Horiguchi et al., 1991; Hu et al., 1994; Makino et al., 1997; Poulos et al., 1986; Romeo et al., 1996; Solliway et al., 1996). The Graziano et al. (1990) and Makino et al. (1997) studies warrant particular attention because of the design (longitudinal), relatively large size (>1000 subjects), and relatively low blood lead levels of the subjects (<40 μg/dL). Both studies support the general conclusion that blood hemoglobin levels are not depressed in association with blood lead concentrations <40 g/dL. In the Kosovo prospective study, no discernable effect of lead on maternal blood hemoglobin levels was evident from a comparison of the high-lead exposure group (mean blood lead 17 μg/dL, range 7–43 μg/dL) with the low lead exposure group (mean blood lead 5.1 μg/dL, range 2–11 μg/dL). Makino et al. (1997) found a positive association between increasing blood lead concentration and increasing blood hemoglobin levels in a longitudinal survey of adult males (n = 1,573) who worked in pigment or vinyl chloride stabilizer manufacture (mean blood lead 13 μg/dL, range 1–39 μg/dL). A simple linear regression model predicted a 1 g/dL increase in blood hemoglobin per 10 μg/dL increase in blood lead concentration (typical level 10–20 g/dL).

Two other cross-sectional studies are also notable, because of design considerations and/or blood lead concentration ranges of the subjects. Solliway et al. (1996) observed no differences in mean blood hemoglobin levels in a comparison of adult male battery manufacture

workers (n = 34, mean blood lead 41 µg/dL, range 23–63 µg/dL) and a matched reference group (n = 56, mean blood lead 7 µg/dL, range 1–13 µg/dL). Hu et al. (1994) conducted a cross-sectional assessment of adult male carpentry workers (n = 119) whose blood lead concentrations were ≤25 µg/dL. Blood hemoglobin was not significantly associated with blood lead concentration. Of note, however, was the finding that increasing patella bone lead was significantly associated with decreasing blood hemoglobin levels. Covariate-adjusted blood hemoglobin levels were predicted to decrease by 1.1 g/dL per 37 µg/g increase (mean of first and fourth quartiles) in patella bone lead.

Studies of lead workers whose blood lead levels were higher than in the studies noted above have, in general, found lower blood hemoglobin levels in association with increasing blood lead concentrations; these include Gennart et al. (1992) with a blood lead range of 40–70 µg/dL, Horiguchi et al. (1991) with a mean blood lead level of 54 µg/dL (SD 16), and Poulos et al. (1986) with mean blood lead range of 21–27 µg/dL. In the latter study (Poulos et al., 1986), blood hemoglobin levels decreased by 0.6–0.9 g/dL per 10 µg/dL increase in blood lead (simple linear regression) in adult males. Analyses or adjustments for potential covariables were not reported for these studies.

6.9.2.3 Erythrocyte Volume and Number

Schwartz et al. (1990) conducted a concentration-response analysis of data collected at the Bunker Hill smelter site in Idaho in 1974, shortly after the failure of the smelter bag house resulted in extensive contamination of the surrounding area with uncontrolled smelter emissions. This analysis is unique in that it collected hematocrit measurements in children (n = 579, age range 1–5 years) who had relatively high blood lead levels (range 11–164 µg/dL, approximately 40% exceeded 40 µg/dL). A logistic model relating blood lead concentration and age to hematocrit predicted a 10% decrease in hematocrit (from 39.5 to 35.5%) in association with blood lead concentrations of 85, 115, and 145 µg/dL at ages 1, 3, and 5 years, respectively (Figure 6-9.2). A 10% probability of anemia (hematocrit <35%) was predicted in association with a blood lead concentration of approximately 20 µg/dL at age 1 year, 50 µg/dL at age 3 years, and 75 µg/dL at age 5 years (Figure 6-9.2).

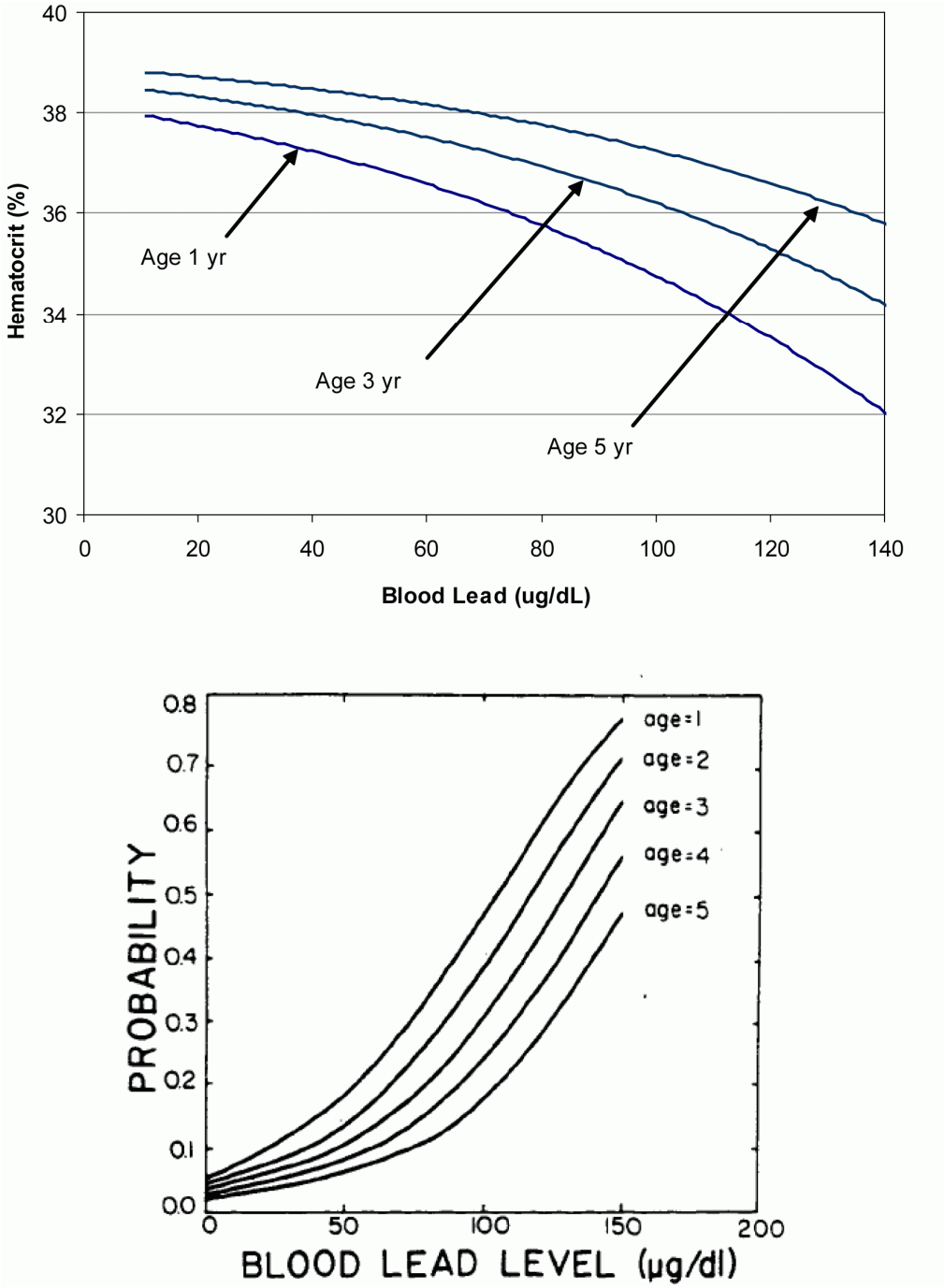

Figure 6-9.2. Relationship between blood lead and hematocrit in children. The top panel shows central tendency predictions based on a logistic regression model relating hematocrit and blood lead concentration, adjusted for age. The regression coefficients relating hematocrit and blood lead were (β = 0.0133 [SE 0.0041], p = 0.0005). The bottom panel shows corresponding concentration-response (hematocrit <35%) relationships.

Source: Schwartz et al. (1990)

Numerous studies of associations between lead exposure and erythrocyte volume (e.g., hematocrit) or number have been reported in adults (Gennart et al., 1992; Horiguchi et al., 1991; Hsiao et al., 2001; Hu et al., 1994; Makino et al., 1997; Osterode et al., 1999; Poulos et al., 1986; Solliway et al., 1996). The Hu et al. (1994) and Makino et al. (1997) studies examined groups of workers that had blood lead concentrations that were relatively low, compared to other studies, and found either no association or weak association between blood lead concentration and hematocrit and/or erythrocyte number. The Hu et al. (1994) cross-sectional study of carpentry workers (n = 119, blood lead concentration range 2–25 μg/dL) found no association between blood lead concentration and hematocrit; however, increasing patella bone lead was associated with a significant decrease in hematocrit. Covariate-adjusted blood hematocrit was predicted to decrease by 0.03% (95% CI: 0.01, 0.05) per 37 μg/g increase (mean of first and fourth quartiles) in patella bone lead. The Makino et al. (1997) longitudinal study of pigment and vinyl chloride stabilizer manufacture workers (n = 1,573; blood lead concentration range 1–39 μg/dL) found a positive association between blood lead concentration and hematocrit, and erythrocyte count. A simple linear regression model predicted an increase in hematocrit of 0.6 (typically 43) and an increase in erythrocyte count of $0.07 \times 10^6/mm^3$ (typically $4–7 \times 10^6/mm^3$) per 10 μg/dL increase in blood lead concentration.

Studies that included subjects who had higher blood lead concentrations (i.e., >40 μg/dL) have, in general, found negative associations between blood lead concentration and hematocrit Gennart et al., 1992; Horiguchi et al., 1991; Poulos et al., 1986; Solliway et al., 1996), with two exceptions, Hsiao et al. (2001) and Osterode et al. (1999). Hsiao et al. (2001) conducted an 11-year retrospective longitudinal analysis of blood lead concentration, hematocrit, and erythrocyte count in a group of battery manufacture workers (n = 30; mean blood lead concentration 30–60 μg/dL). A repeated measures regression analysis (generalized estimation equation) yielded a significant association between increasing blood lead concentration and increasing hematocrit and erythrocyte count. Osterode et al. (1999) measured erythrocyte number and packed cell volume in a group of lead workers (n = 20) and an age-matched reference group (n = 20). Mean blood lead concentration was 45.5 μg/dL (range 16–91 μg/dL) in the lead workers and 4.1 μg/dL (range 3-14 μg/dL) in the reference group. Mean erythrocyte number and packed cell volume in the lead workers and reference group were not different.

6.9.2.4 Erythropoiesis

Several studies have found associations between lead exposure and serum erythropoietin levels in children (Graziano et al., 2004; Liebelt et al., 1999) and adults (Graziano et al., 2001; Osterode et al., 1999; Romeo et al. 1996). A qualitative summary of outcomes from these studies are provided in (Table 6-9.3).

Two studies have examined possible association between lead exposure and serum erythropoietin levels in children. In the Kosovo prospective study (Factor-Litvak et al., 1999, 1998; Graziano et al., 2004) a significant association was evident between increasing blood lead concentration (3–70 μg/dL) and increasing serum erythropoietin levels after adjustment for age and blood hemoglobin levels (Figure 6-9.3). The association weakened with age; it was significant at ages 4.5 and 6.5 years, but not at ages 9.5 or 12 years. A multivariate linear regression model predicted a 36% increase in serum erythropoietin per 10 μg/dL increase (3-13 μg/dL, hemoglobin 13 g/dL) in blood lead at age 4.5 years, and an 18% increase per 10 μg/dL at age 6.5 years. These outcomes suggest that erythropoiesis is stimulated in children in association with increasing blood lead concentrations below 40 μg/dL and in the absence of depressed blood hemoglobin levels.

A smaller cross-sectional study examined serum erythropoietin levels in a group of children (n = 89), 1 to 6 years of age (Liebelt et al., 1999). The blood lead concentration range in the study group (2–84 μg/dL) was similar to that in the Graziano et al. (2004) study and, consistent with this study, Liebelt et al. (1999) found no association between blood lead concentration and serum hemoglobin levels. However, in contrast to the Graziano et al. (2004) study, blood hemoglobin-adjusted serum erythropoietin levels decreased in association with an increase in blood lead concentration (0.3 mIU/mL decrease per 10 μg/dL increase blood lead). The Liebelt et al. (1999) study did not include age as a covariate in the regression model, which was shown in the Kosovo prospective study to be a significant covariable in blood lead-serum erythropoietin relationship (Graziano et al., 2004); this may have contributed to the different outcome in the two studies. Liebelt et al. (1999) studied a convenience sample from a lead/primary care clinic (rather than a prospectively selected cohort) that specifically excluded children who had symptoms of severe iron deficiency, or were taking iron supplements or other bone marrow suppressing drugs. Iron status of the children in the Graziano et al. (2004) study was not reported. However, serum ferritin levels in the mothers, at mid-pregnancy, was not

Table 6-9.3. Summary of Results of Selected Studies of Associations Between Lead Exposure and Serum Erythropoietin

Study	Subjects	n[a]	Blood Lead (µg/dL)		Serum Erythropoietin	Comment
			Mean (SD)	Range		
Children						
Graziano et al. (2004)	ages: 4.5–12 yr	311	6–9, 31–39[b]	3–70	+	adjusted for age, blood Hgb
Liebelt et al. (1999)	ages: 1–6 yr	86	18[c]	2–84	–	adjusted for blood Hgb
Adults						
Graziano et al. (1990)	pregnant women	48	NR	2–40	–	stratified by blood Hgb
Osterode et al. (1999)	male lead workers	40	45	16–91	–	adjusted for blood PCV
Romeo et al. (1996)	male lead workers	141	30, 65[b,d]	30–92	–	no association with blood Hgb

–, decrease; +, increase; Hgb, hemoblogin; NR, not reported; PCV, packed cell volume SD, standard deviation.

[a] total number of subjects (including reference group)
[b] range of means of low and higher exposure groups
[c] median
[d] reference group mean was 10 µg/dL (range 3–20)

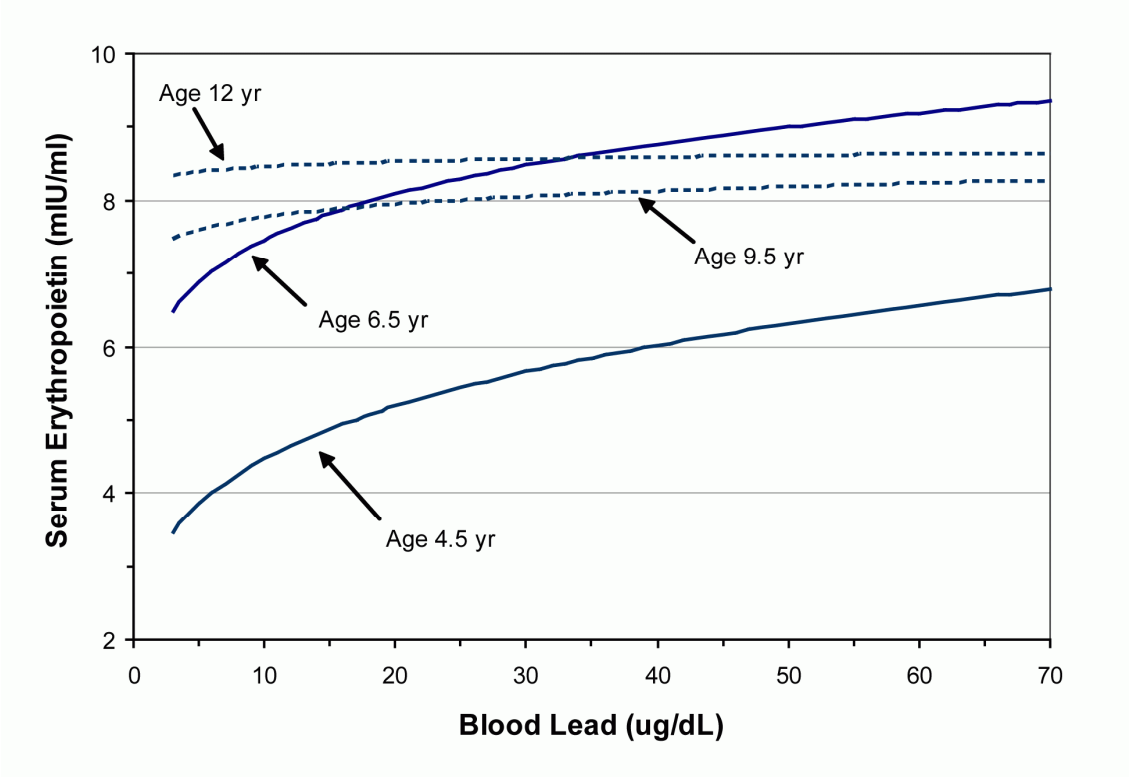

Figure 6-9.3. Relationship between blood lead and serum erythropoietin in children. Shown are central tendency predictions based a generalized estimating equation (for repeated measures) relating serum erythropoietin and cumulative lifetime average blood lead concentration, adjusted for age and blood hemoglobin levels (13 g/dL). The regression coefficients relating erythropoietin and blood lead were significant for ages 4.5 (β = 0.21 [95% CI: 0.13, 0.30], p < 0.0001) and 6.5 years (β = 0.12 [95% CI: 0.03, 0.20], p < 0.001).

Source: Graziano et al. (2004).

indicative of iron deficiency (Graziano et al., 1990). Although the direction of the outcome measure was different in the two studies, both studies (Graziano et al., 2004; Liebelt et al., 1999) found evidence for an effect of lead exposure on serum erythropoietin levels in the absence of significant lead-associated changes in blood hemoglobin levels.

Three studies have found associations between lead exposure and changes in erythropoiesis biomarkers in adults. As part of the Kosovo prospective study, serum erythropoietin was measured at mid-pregnancy and at term in a subset of women enrolled in the study (Graziano et al., 1991). The high- and low-lead cohorts were constructed from the six

highest and lowest mid-pregnancy blood lead concentrations, within each of four blood hemoglobin strata, ranging from 9.0 to 12.9 g/dL. Mean blood lead concentrations in the strata ranged from 17 to 39 µg/dL in the high-lead group and 2.4 to 3.6 µg/dL in the low lead group. Serum erythropoietin levels significantly decreased in association with increasing blood lead concentration, independently of an effect of blood hemoglobin (Figure 6-9.4). Romeo et al. (1996) also found an association between increasing blood lead concentration and decreasing serum erythropoietin, in the absence of discernable changes in blood hemoglobin levels, in a comparison of groups male lead workers (n = 28, blood lead range 30–92 µg/dL) and a similar-aged reference group (n = 113, mean blood lead 10 µg/dL, range 3–20). Osterode et al. (1999) examined several measures of erythropoiesis in a group of lead workers (n = 20, mean age 46 years) and in an age-matched reference group (n = 20). Mean blood lead concentration was 45.5 µg/dL (range 16–91) in the lead workers and 4.1 µg/dL (range 3–14) in the reference group. Mean blood hemoglobin levels in the lead worker and reference groups were not different. Lead workers with had blood lead concentrations ≥60 µg/dL had significantly lower circulating erythrocyte progenitor cells than the reference group. Also, erythrocyte progenitor cell number was significantly negatively correlated with blood lead concentration and urine lead concentration. Serum erythropoietin levels increased exponentially with decreasing packed blood cell volume in the reference group, but not in the lead workers (i.e., serum erythropoietin level was not significantly correlated with packed cell volume in the lead workers). Thus, unlike the reference group (blood lead concentration ≤14 µg/dL), lead workers appeared to have a suppressed erythropoietin response to declining blood cell volume.

Collectively, the results of the above studies suggest that lead exposure depresses serum erythropoietin levels, in the absence of significant depression in blood hemoglobin levels. Lead-induced nephrotoxicity may contribute to a suppression of erythropoietin levels in lead-exposed individuals. Although this cannot be entirely ruled out in these studies, both the Romeo et al. (1996) and Osterode et al. (1999) studies excluded people who had a history of hematological or kidney disease. Nevertheless, renal nephrotoxicity, including proximal tubular nephropathy, could have been a confounder in these studies which included subjects whose blood lead concentrations were >40 µg/dL.

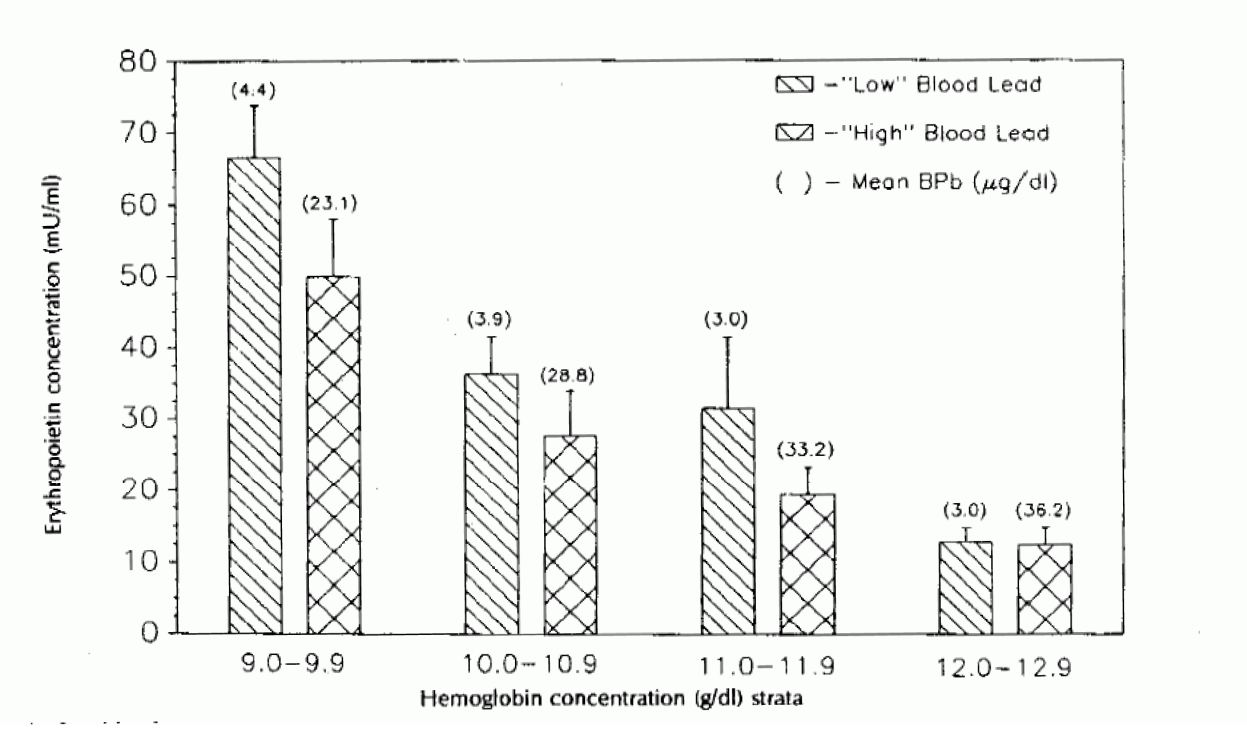

Figure 6-9.4. Association between blood lead concentration and serum erythropoietin in pregnant women. Shown are combined data for mid-pregnancy and delivery. Each bar represents the mean (±SD) of 12 subjects. ANOVA of the data at mid-pregnancy and at delivery showed blood lead effects (p = 0.049, p = 0.055, respectively) and blood hemoglobin effects (p = 0.0001. p = 0.009, respectively), with no significant interaction between the two variables.

Source: Graziano et al. (2001).

6.9.2.5 Other Effects on Erythrocyte Metabolism and Physiology

6.9.2.5.1 Erythrocyte Nucleotide Metabolism

Lead inhibits erythrocyte pyrymidine-5′nucleotidase (P5N) and adenine dinucleotide synthetase (NADS). Associations between increasing blood lead concentration and decreasing blood P5N and NADS activity have been observed in studies of lead workers (Kim et al. 2002; Mohammed-Brahim et al., 1985; Morita et al., 1997). Mean blood lead concentrations in these study groups were ≥35 μg/dL and ranged up to 80 μg/dL.

6.9.2.5.2 Erythrocyte Deformability

Horiguchi et al. (1991) compared the deformability of erythrocytes collected from adult male secondary lead refinery workers (n = 17, age range 24–58 years) with a reference group of male subjects (n = 13, age range 22–44 years). Erythrocyte deformability was assessed as microfilterability of erythrocytes under a negative (−10 cm H_2O) pressure head. Erythrocytes from the lead workers showed significantly lower deformability compared to the reference group. The mean blood lead concentration in the lead workers was 53.5 µg/dL (SD 16.1).

6.9.2.5.3 Erythrocyte Membrane Transport

Hajem et al. (1990) measured erythrocyte membrane activities of Na^+-K^+-ATAase, Na^+-K^+-co-transport, Na^+-Li^+-antiport, and passive Na^+ and K^+ permeability in erythrocytes collected from adult males (n = 122, geometric mean blood lead: 16 µg/dL, range 8.0–33.0) and hair lead was 5.3 µg/g (95% CI: 4.44, 6.23, range 0.9–60). Na^+-K^+-co-transport activity was negatively correlated with blood lead concentration but not with hair lead (geometric mean 5.3 µg/g, range 0.9–60), and Na^+-K^+-ATPase activity was negatively correlated with hair lead, but not with blood lead.

6.9.3 Effects of Lead on the Endocrine System

6.9.3.1 Summary of Key Findings of the Effects of Lead on the Endocrine System from the 1986 Lead AQCD

The 1986 Lead AQCD concluded that various endocrine processes may be affected by lead at relatively high exposure levels. These included effects on thyroid hormone levels (e.g., Refowitz, 1984; Robins et al., 1983), effects on male sex hormone levels (e.g., Braunstein et al., 1978), and impairment of the production of 1,25-dihydroxy vitamin D (1,25-OH-D) (e.g., Rosen et al., 1980). Effects on these endocrine systems were concluded to be apparent only at blood lead concentrations exceeding 30–40 µg/dL. The 1986 Lead AQCD concluded that studies from which the effects of lead on reproductive hormones in females could be assessed were lacking.

More recent epidemiologic studies have examined possible associations between lead exposure (as reflected by blood and/or bone lead levels) and various biomarkers of endocrine function, including the thyroid, male reproductive, and calcitropic endocrine systems. These studies have examined endocrine outcomes at lower blood lead ranges and in the absence of

overt clinical lead toxicity, and have more rigorously attempted to control for confounding factors. Evidence for lead effects on these systems, in association with blood lead concentrations below 30–40 μg/dL, remains absent. The strongest study designs have yielded no associations, or weak associations, between lead exposure and thyroid hormone status (Erfurth et al., 2001; Schumacher et al., 1998; Tuppurainen et al., 1988; Zheng et al., 2001). Similarly, studies of the male reproductive system that attempted to control for confounding effects of age, have yielded mixed outcomes (Alexander et al., 1998, 1996; Erfurth et al., 2001; Gustafson et al., 1989; McGregor and Mason, 1990; Ng et al., 1991). Results of a more recent epidemiologic study of the calcitropic endocrine system in children suggest that associations between serum vitamin D status and blood lead may not be present in calcium-replete children who have average lifetime blood lead concentrations below 25 μg/dL (Koo et al., 1991). In adults, exposures to lead that result in blood lead concentrations >40–60 μg/dL may increase, rather than decrease, circulating levels of 1,25-OH-D and PTH (Kristal-Boneh et al., 1999; Mason et al., 1990), possibly as a compensatory response to increased urinary calcium losses, secondary to impaired kidney function. Details regarding the design of these studies and outcomes are presented in Annex Tables AX6-9.5 and AX6-9.6. Outcomes of the most pertinent studies are summarized below.

6.9.3.2 Thyroid Endocrine Function

Several studies have examined possible associations between lead exposure and thyroid hormone status. Most of these have been studies of occupational exposures. The results of these studies have been mixed; some studies have found significant associations with lead exposure (e.g., blood lead concentration), but most studies have found none or relatively weak associations. In studies that have controlled for the effects of age, outcomes also have been mixed, with the strongest study designs finding none or weak associations between lead biomarkers and thyroid hormone status (Erfurth et al., 2001; Schumacher et al., 1998; Tuppurainen et al., 1988; Zheng et al., 2001). The strength of the association and, possibly, the direction of the effect (i.e., increase or decrease in hormone levels) may change with exposure duration or level (Robins et al., 1983; Tuppurainen et al., 1988). The overall picture that emerges is that those studies that have included subjects having blood lead concentrations exceeding 100 μg/dL have found depression of serum T3 and/or T4 levels, without a detectable increase in serum TSH. However, studies in which the blood lead distribution was dominated by

levels well below 100 µg/dL, have found either no effects or subclinical increases in serum T3, T4, with no change in TSH levels. Outcomes from the most pertinent studies are summarized qualitatively in Table 6-9.4 and are described in greater detail below.

Siegel et al. (1989) measured serum total thyroxine (TT4) and free thyroxine (FT4) in children ages 11 months to 7 years (n = 68) who were outpatients at a clinical care facility. Mean blood lead concentration in the study group was 25 µg/dL (range 2–77). In a simple (univariate) linear regression analysis, hormone levels were not significantly associated with blood lead concentration.

Zheng et al. (2001) measured concentrations of TT4 and transthyretin (TTR) in serum and cerebral spinal fluid (CSF) of adult hospital patients (n = 82) admitted for evaluation of CSF clinical chemistry (e.g., for head wounds, tumors, neurological symptoms). Mean blood lead concentration was 14.9 µg/dL (SD 8.3). Age-adjusted serum TT4 and TTR, and CSF TT4 were not significantly associated with blood lead concentration; however, increasing CSF lead concentration was associated with decreasing CSF TTR levels ($r = -0.30$, $p = 0.023$).

Possible associations between lead exposure and thyroid hormone status have been examined in several studies of lead workers (Dursun and Tutus, 1999; Erfurth et al., 2001; Gennart et al., 1992; Gustafson et al., 1989; Horiguchi et al., 1987; Lopez et al., 2000; Refowitz, 1984; Robins et al., 1983; Schumacher et al., 1998; Singh et al., 2000; Tuppurainen et al., 1988). Of these, six warrant particular attention because the design and/or analysis attempted to control for effects of age (Erfurth et al., 2001; Dursun and Tutus, 1999; Gustafson et al., 1989; Schumacher et al., 1998; Tuppurainen et al., 1988; Robins et al., 1983). Outcomes of these studies are summarized in Table 6-9.4. The largest studies were Erfurth et al. (2001), Schumacher et al. (1998), and Tuppurainen et al. (1988).

Erfurth et al. (2001) was a cross-sectional study of secondary smelter workers (n = 62) and a reference group of metal (not lead) workers (n = 26). Excluded from the study were individuals with ongoing thyroid disease or who were taking thyroid hormone supplements or other drugs that would interfere with thyroid hormone levels (e.g., beta-blockers). Median blood lead concentration in the lead workers was 31 µg/dL (range 8–93 µg/dL). Age-adjusted basal serum levels of FT3, FT4, and TSH were not associated with blood, urine, or finger bone lead levels. Thyroid releasing hormone (TRH)-induced TSH secretion (area under serum TSH concentration-time curve) was measured in an age-matched subset of the study group (9 lead

Table 6-9.4. Summary of Results of Selected Studies of Associations Between Lead Exposure and Thyroid Hormone Levels

Study	Subjects	n[a]	Blood Lead (µg/dL)		T3	T4	TSH
			Mean (SD)	Range			
Children							
Siegel et al. (1989)	children, 11 mo–7 yrs	68	25	2–77	NR	o	NR
Adults							
Dursun and Tutus (1999)	metal powder manufacture workers	57	17.1 (9.0)	1–36	+	+	o/o[b]
Erfurth et al. (2001)	secondary smelter workers	88	31.1[c]	4–93	o	o	o
Gustafson et al. (1989)	secondary smelter workers	42	39.4 (2.1)	NR	o	+	o
Robins et al. (1983)	brass foundry workers	47	NR	16–127	NR	–	NR
Schumacher et al. (1998)	primary smelter workers	151	24.1	15>40%	o	o	o
Tuppurainen et al. (1988)	battery manufacture workers	176	55.9 (23.8)	5–134	–	–	o
Zheng et al. (2001)	general population	82	14.9 (8.3)	NR	NR	o	NR

–, decrease; +, increase; o, no effect; NR, not reported; T3, triiodothyronine; T4, thyroxine; TSH, thyroid stimulating hormone

[a] Total number of subjects (including reference group)
[b] basal/thyroid releasing hormone-stimulated
[c] median

workers and 11 reference subjects) and was not significantly different in the two groups. The Schumacher et al. (1998) study measured serum FT4, TT4, and TSH levels in a group of male workers (n = 151) at the Trail British Columbia smelter complex. Excluded from the study were individuals who had ongoing clinical thyroid disease. Mean blood lead concentration in the study group was 24 μg/dL (15% >40 μg/dL). Covariate-adjusted (age, alcohol consumption) hormone levels were not significantly associated with current blood lead concentration or 10-year average blood lead concentrations. Prevalence of abnormal hormone values was also unrelated to blood lead concentration.

Tuppurainen et al. (1988) measured serum total triiodothyronine (TT3), FT4, TT4, and TSH levels in a group of male battery manufacture workers (n = 176). Mean blood lead concentration was 56 μg/dL (range 14–134 μg/dL). Although, hormone levels were not significantly associated with blood lead concentrations, increasing exposure (i.e., employment) duration was significantly associated with decreasing FT4 ($r^2 = 0.071$, $p = 0.001$) and TT4 ($r^2 = 0.059$, $p = 0.021$) levels. The r^2 was not improved by including age or blood lead as covariables. Strength of the association was greater when the analysis was restricted to workers who had an exposure duration >7.6 years (FT4: $r^2 = 0.33$, $p < 0.002$; TT4: $r^2 = 0.21$, $p < 0.001$). Consistent with the results of the Tuppurainen et al. (1988) study, Robins et al. (1983) found a significant association between increasing blood lead concentration and decreasing FT4 ($r^2 = 0.085$, $p = 0.048$) in a group of brass foundry workers (n = 47). The blood lead range in the subjects was 16–127 μg/dL. When stratified by race (black, white) the association was significant in the black stratum ($r^2 = 0.21$, $p = 0.03$), but not in the white stratum ($r^2 = 0.05$, $p = 0.27$). The strength of association was not changed by including age in the regression model. Both the Robins et al. (1983) and Tuppurainen et al. (1988) included subjects with blood lead concentrations >100 μg/dL.

Blood lead concentrations were lower in the Dursun and Tutus (1999) and Gustafson et al. (1989) studies than in the above studies, and both studies found significant associations between lead exposure and increasing serum TT4 levels. Dursun and Tutus (1999) measured serum FT3, TT3, FT4, TT4, and TSH in a group of metal powder manufacture workers (n = 27) and a reference group (n = 30). Mean blood lead concentration in the workers was 17 μg/dL (range 9–36 μg/dL). A linear regression model that included age, blood lead concentration, and exposure duration, indicated a significant association between increasing exposure duration and increasing

serum TT4 levels ($r^2 = 0.3$, $p = 0.03$). The Gustafson et al. (1989) study examined a group of male secondary smelter workers (n = 21) and reference subjects, individually matched to the lead workers by age, sex, and work shift. Mean blood lead concentration in the workers was 39 µg/dL (SD 2). Serum TT4 levels were significantly higher ($p < 0.02$) in the lead workers compared to the reference group. The difference strengthened when the analysis was restricted to the age range <40 years ($p = 0.01$).

6.9.3.3 Reproductive Endocrine Function

6.9.3.3.1 Male Reproductive Endocrine Function

Low testosterone (TES) levels, blunted sex hormone secretion in response to gonadotropin releasing hormone (GNRH), and defects in spermatogenesis have been observed in humans exhibiting clinical neurological symptoms of lead poisoning (Braunstein et al., 1978; Cullen et al., 1984). However, the effects of lower exposure levels on reproductive endocrine status are less clear. Possible associations between lead exposure and changes in male reproductive hormone levels have been examined in studies of lead workers. Of these, five studies attempted to control for effects of age, an important determinant of testosterone levels (Alexander et al., 1998; Erfurth et al., 2001; Gustafson et al., 1989; McGregor and Mason, 1990; Ng et al., 1991). The outcomes from these studies are qualitatively summarized in Table 6-9.5. Blood lead ranges in the latter studies were similar (4–90 µg/dL), yet outcomes were mixed, with no change (Erfurth et al., 2001; Gustafson et al., 1989; McGregor and Mason, 1990) or subclinical decrease (Alexander et al., 1998, 1996; Ng et al., 1991) in serum testosterone (TES) in association with lead exposure. Mixed effects were observed for the effect of lead exposure on serum follicle stimulating hormone (FSH) and luteinizing hormone (LH), increases (McGregor and Mason, 1990; Ng et al., 1991), decreases (Gustafson et al., 1989), and with no change (Alexander et al., 1998, 1996; Erfurth et al., 2001) in hormone levels observed. The inconsistency in the direction of effects on TES and the two androgen regulating pituitary hormones, FSH and LH, is particularly noteworthy. In the absence of an abnormality in the hypothalamic-pituitary regulation of testosterone levels, an association between declining serum TES (or free TES) and increasing FSH and LH levels would be expected. Erfurth et al. (2001) observed a suppressed FSH response to GNRH in a group of lead workers compared to an

Table 6-9.5. Summary of Results of Selected Studies of Associations Between Lead Exposure and Male Sex Hormone Levels in Adults

Study	Subjects	n[a]	Blood Lead (µg/dL)		FSH	LH	PRL	TES
			Mean (SD)	Range				
Alexander et al. (1998, 1996)	primary smelter workers	152	NR	5–58	o	o	NR	–[b]
Erfurth et al. (2001)	secondary smelter workers	88	31.1[c]	4–93	o/-[d,e]	o/o[d]	o/o[d]	o[d]
Gustafson et al. (1989)	secondary smelter workers	42	39.4 (2.1)	NR	–	–	o	o
McGregor and Mason (1990)	lead workers	176	NR	17–77	+	+	NR	o
Ng et al. (1991)	battery manufacture workers	171	35 (13)	10–72	+	+	o	–

–, decrease; +, increase; o, no effect; NR, not reported, FSH, follicle stimulating hormone, LH, luteinizing hormone; PRL, prolactin; TES, testosterone

[a] total number of subjects (including reference group)
[b] in association with increasing semen lead levels, not with blood lead
[c] median
[d] basal/gonadotropin releasing hormone-stimulated
[e] effect was evident in comparison between groups, but not in multivariate regression that adjusted for age

age-matched reference group; however, the magnitude of the response was not significantly associated with lead exposure indices in a multivariate regression analysis that accounted for age.

Alexander et al. (1998, 1996) examined serum FSH, LH, and TES in males (n = 152) who worked at the Trail British Columbia smelter complex. Covariate-adjusted hormone levels and prevalence of clinically abnormal values were unrelated ($p \geq 0.05$) to blood lead concentration (range 5–58 µg/dL); however, increasing semen lead concentration (range 0.3-17 µg/dL) was significantly associated with decreasing semen testosterone levels ($p = 0.004$). Erfurth et al. (2001) measured serum TES, sex hormone binging globulin (SHBG), and GNRH-stimulated changes in serum FS, LH, and PRL in male secondary smelter workers (n = 62) and in a reference group (n = 26). Mean blood lead in the lead workers was 31 µg/dL (range 8-93 µg/dL). Age-adjusted basal hormone levels were unrelated to blood, plasma, or urine lead concentrations. In an age-matched subset of the cohorts (n = 9 lead workers, n = 11 reference), median GNRH-stimulated serum FSH was significantly lower in lead workers than in the reference group; however, GNRH-stimulated LH, FSH, and PRL were not significantly associated with any of the lead measures in a multivariate regression analysis. Gustafson et al. (1989) measured serum FSH, LH, and TES (total and free) in a group of male secondary smelter workers (n = 21) and in a group of reference subjects individually matched to the lead workers by age, sex, and work shift. Mean blood lead concentrations were 39 µg/dL (SD 2) in the lead workers and 5.0 µg/dL (SD 0.2) in the reference group. Serum FSH levels were significantly lower ($p = 0.009$) in lead workers compared to reference group. When the analysis was restricted to the age range <40 years, lead workers had significantly lower FSH and LH compared to the reference group. McGregor and Mason (1990) measured serum FSH, LH, TES, and SHBG in a group of male lead workers (n = 90) and in a reference group (n = 86). Blood lead range in the lead workers was 17–77 µg/dL; blood lead concentrations in the reference subjects were <12 µg/dL. Prevalences of abnormal hormone levels in the lead workers and reference group were not different; however, age-adjusted serum FSH was significantly higher in lead workers compared to reference group and increasing FSH levels were significantly associated with increasing blood lead concentrations. Increasing serum LH was significantly associated with increasing exposure duration but not with blood lead concentration or age. Serum TES or SHBG levels were unrelated to blood lead concentration or exposure duration. Ng et al. (1991) measured serum FSH, LH, PRL, and TES in a group of male battery

manufacture workers (n = 122) and a reference group (n = 49). Mean blood lead concentrations were 35 µg/dL (range 10–77 µg/dL) in the lead workers and 8 µg/dL (range 3-15 µg/dL) in the reference group. When cohorts were stratified by age, serum FSH and LH levels were significantly higher in lead workers <40 years of age compared to corresponding age stratum of the reference group; serum TES was significantly lower in lead workers ≥40 years of age, compared to the same age stratum in the reference group. Covariate-adjusted (age, tobacco smoking) serum TES levels were significantly lower in lead workers in the 10-year exposure duration stratum, compared to the reference group. Covariate-adjusted serum FSH and LH were significantly higher in lead workers in the <10-year exposure duration stratum, compared to the reference group.

6.9.3.3.2 Female Reproductive Endocrine Function

Although delays in sexual maturation in humans have been associated with blood lead concentrations (Selevan et al., 2003; Wu et al., 2003), and lead has been shown to alter levels of female sex hormones and the menstrual cycle in nonhuman primates (Foster, 1992; Franks et al., 1989; Laughlin et al., 1987), epidemiologic studies of interactions between lead exposure and reproductive endocrinology in females have not been reported. Lead introduced into cultures of human ovarian granulosa cells suppresses progesterone production (Paksy et al., 2001) and suppresses expression of aromatase and estrogen receptor β (Taupeau et al., 2003).

6.9.3.4 Pituitary and Adrenal Endocrine Function

Several studies of possible associations between lead exposure and levels of pituitary hormones that regulate production and secretion of thyroid hormones (see Section 6.9.3.2) and reproductive hormones (see Section 6.9.3.3) have been reported. In addition to the above studies, Gustafson et al. (1989) found that serum cortisol levels were lower in a group of male secondary smelter workers (n = 21) compared to a reference group individually matched to the lead workers by age, sex, and work shift. Mean blood lead concentration were 39 µg/dL (SD 2) in the workers and 5.0 µg/dL (SD 0.2) in the reference group. Campbell et al. (1985) measured various biomarkers of status of the renin-angiotensin-aldosterone system in male welders (n = 5) and reference subjects (n = 8). Mean blood lead concentration was 35 µg/dL (range 8-62 µg/dL). Significant positive correlations were observed between blood lead concentration and plasma

aldosterone ($r = 0.53$, $p < 0.002$), which may have been, at least in part, secondary to a lead effect on plasma renin activity ($r = -0.76$, $p < 0.001$) and angiotensin I levels ($r = 0.68$, $p < 0.002$). Saenger et al. (1984) found lower urinary levels of 6-β-OH-cortisol, but not cortisol, in children who had elevated urinary lead in an EDTA provocation test (>500 μg/24 h), compared to children who did not have elevated urinary lead levels, or whose blood lead concentrations were <30 μg/dL. The change in urinary excretion of 6-β-OH-cortisol in the absence of a change in cortisol levels may reflect an effect of lead on liver cytochrome P450 activity, rather than an effect on the adrenal gland (see Section 6.9.4).

6.9.3.5 Calcitropic Endocrine Function

Children exposed to relatively high level of lead >30 μg/dL may exhibit depressed levels of circulating 1,25-OH-D (Mahaffey et al., 1982; Rosen et al., 1980). These effects were not detected in a study of calcium-replete children with average lifetime blood lead levels below 25 μg/dL (Koo et al., 1991). In adults, lead exposures that result in blood lead concentrations >40–60 μg/dL may increase, rather than decrease, circulating levels of 1,25-OH-D and PTH. These studies also are summarized in Annex Tables AX6-9.5 and AX6-9.6. Outcomes from the more pertinent studies are qualitatively summarized in Table 6-9.6 and are discussed in greater detail below.

Epidemiologic studies of possible associations between lead exposure and vitamin D status in children have yielded mixed results. Mahaffey et al. (1982) and Rosen et al. (1980) observed lower 1,25-OH-D in association with increasing blood lead concentration. Koo et al. (1991) found no association between 1,25-OH-D and blood lead concentration. The Koo et al. (1991) study was a longitudinal analysis of a subset of a prospective study of pregnancy outcomes. Serum calcium magnesium, phosphorus, PTH, CAL, 25-OH-D, 1,25-OH-D, and bone mineral content were measured in children ($n = 105$) at ages 21, 27, and 33 months. Mean lifetime average blood lead concentrations (based on quarterly assessments) was 9.7 μg/dL (range 4.8–23.6 μg/dL). The range of highest values observed was 6–63 μg/dL. A structural equation model was developed that initially considered age, sex, race, sampling season, and dietary intake of calcium, phosphorus, and vitamin D as covariables; the final model retained age, sex, race, and sampling season. Decreasing blood lead (ln-transformed) was significantly associated with covariate-adjusted decreasing serum phosphorus. No other covariate-adjusted

Table 6-9.6. Summary of Results of Selected Studies of Associations Between Lead Exposure and Calcitropic Hormones

Study	Subjects	n[a]	Blood Lead (µg/dL)		PTH	CAL	1,25D	25D
			Mean (SD)	Range				
Children								
Koo et al. (1991)	ages: 21, 27, 33 mo	105	9.7	5–24	o	o	o	o
Mahaffey et al. (1982)	ages: 1–16 yr	177	NR	12–120	o	o	–	o
Rosen et al. (1980)	ages: 1–5 yr	45	18, 47, 74[b]	10–120	+	o	–	–
Adults								
Chalkley et al. (1998)	smelter workers[c]	19	47	21–76	NR	NR	+[c]	o
Kristal-Boneh et al. (1998)	battery manufacture workers	140	43	1–77	+	NR	+	NR
Mason et al. (1990)	lead workers	138	NR	15–95	o	NR	+	NR

–, decrease; +, increase; o, no effect; NR, not reported, PTH, parathyroid hormone; CAL, calcitonin; 1,25D, 1,25-dihydroxyvitamin D; 25D, 25-hydroxyvitamin D

[a] total number of subjects (including reference group)
[b] group means: low, moderate, high
[c] cadmium, lead, zinc smelter workers, effect on 1,24D in association with high blood cadmium and lead and high urinary cadmium

outcomes were significantly associated with blood lead. The distribution of dietary calcium intakes was 4% for ≤600 mg/day, 55% for 600–1200 mg/day, and 41% for >1200 mg/day. Intakes of phosphorous were similar, suggesting that the subjects were nutritionally replete with respect to these two nutrients.

The different outcomes in Koo et al. (1991) compared to the Mahaffey et al. (1982) and Rosen et al. (1980) studies may reflect, in part, the lower blood lead range in the subjects in Koo et al. (1991) (range of lifetime average 5–24 µg/dL, range of observed highest values 6–63 µg/dL) compared to the Mahaffey et al. (1982) and Rosen et al. (1980) studies (10–120 µg/dL). Subjects in the Koo et al. (1991) study also had higher calcium intakes (4% with ≤600 mg/day, 43% with >1200 mg/day) than in the Rosen et al. (1980) study (mean 580 mg/day [SE 15] in high blood lead group). Calcium intake (and/or related nutritional factors) may also have been an uncontrolled confounder in the Rosen et al. (1980) study, as higher blood lead concentration appeared to be associated with lower calcium intakes (Sorrell et al., 1977). Mahaffey et al. (1982) did not report calcium intakes. Thus, the effect of lead exposure on vitamin D status may be more pronounced at higher blood lead concentrations (i.e., >60 µg/dL) and in combination with lower intakes of calcium (or other nutritional limitations).

Studies of lead workers have found evidence for higher serum levels of 1,25-OH-D and PTH in association with increasing blood lead concentration (Chalkley et al., 1998; Kristal-Boneh et al., 1998; Mason et al., 1990). The Chalkey et al. (1998) study was a small study (n = 19) of subjects exposed to both cadmium and lead, and effects of lead and cadmium on 1,25-OH-D could not be isolated. The Kristal-Boneh et al. (1998) and Mason et al. (1990) studies included larger samples of subjects whose exposure was primarily, but not exclusively, to lead. Attempts were made to control for effects of age and, in the Kristal-Boneh et al. (1998) study, other potential covariables. Kristal-Boneh et al. (1998) measured serum calcium, magnesium, phosphorus, PTH, 25-OH-D, and 1,25-OH-D in a group of male battery manufacture workers (n = 56) and a reference group (n = 90). Mean blood lead concentrations were 43 µg/dL (SD 14, range 1-77 µg/dL) in the lead worker group and 4.5 µg/dL (SD 2.6, range 1.4–19 µg/dL) in the reference group. Serum 1,25-OH-D and PTH, but not 25-OH-D, were significantly higher in lead workers compared to the reference group. Increasing blood lead concentration (ln-transformed) was significantly associated with covariate-adjusted increasing serum PTH and 1,25-OH-D levels. No effects on serum calcium were apparent. Occupational

lead exposure was also significantly associated with increasing PTH and 1,25-OH-D level. Covariates retained in the multivariate model were age, alcohol consumption, smoking; calcium intake, magnesium intake, and calorie intake. Mason et al. (1990) measured serum calcium, phosphate, PTH, and 1,25-OH-D in male lead workers (n = 63) and in a reference group (n = 75) and found significantly higher prevalence of elevated 1,25-OH-D (defined as >2 SD higher than reference mean) in lead workers (13%) compared to the reference group (1.3%). Serum levels of 1,25-OH-D were also significantly higher in lead workers compared to the reference group. After stratification of the lead workers into exposure categories (high exposure: blood lead ≥40 μg/dL and bone lead ≥40 μg/g; low exposure: blood lead ≤40 μg/dL and bone lead ≤40 μg/g), serum 1,25-OH-D levels were significantly higher in the high lead group. Serum calcium levels were not different in the two groups. Increasing blood lead was significantly associated with increasing 1,25-OH-D levels ($r^2 = 0.206$; with age and bone lead included, $r^2 = 0.218$). After excluding 12 subjects whose blood lead concentrations >60 μg/dL, the regression coefficient was no longer significant ($r^2 = 0.162$, $p = 0.26$).

6.9.4 Effects of Lead on the Hepatic System

6.9.4.1 Summary of Key Findings of the Effects of Lead on the Hepatic System from the 1986 Lead AQCD

The 1986 Lead AQCD noted that effects of lead on liver function in humans had not been extensively studied. Possible association between lead exposures (blood lead concentrations >70 μg/dL) and nonspecific liver injury (i.e., increases in liver enzymes in serum) were noted based on studies of workers (e.g., Cooper et al., 1973; Hammond et al., 1980). Also noted was evidence for possible association of suppression of hepatic cytochrome P450 activity with high blood lead concentrations (>70 μg/dL) (Meredith et al., 1977).

Few studies of hepatic effects of lead on humans have been reported since the 1986 Lead AQCD. Studies of hepatic enzyme levels in serum suggest that liver injury may be present in lead workers; however, associations specifically with lead exposures are not evident (Al-Neamy et al., 2001; Hsiao et al., 2001). Studies of urinary metabolites of cytochrome P450 phenotypes CYP2A6 and CYP3A4 suggest possible associations between lead exposure and suppression of hepatic enzyme activity. The effect on CYP2A6 activity was observed in children with high lead burdens (i.e., blood lead concentration >40 μg/dL, EDTA-provoked urinary lead >500 μg/dL).

The effect on CYP3A4 was observed in association with blood lead ranges of approximately 30-112 μg/dL (based on reported serum lead concentrations). These studies are summarized in Annex Table AX6-9.7 and the most pertinent findings are discussed below.

6.9.4.2 Non-specific Hepatic Injury

Possible association between occupational lead exposure and liver injury has been assessed from measurements of serum enzymes (Al-Neamy et al., 2001; Hsiao et al., 2001). Al-Neamy et al. (2001) found significantly higher serum activity of alkaline phosphatase (AP) and lactate dehydrogenase (LDH), both within clinically normal ranges, in a group (n = 100) of male lead workers (e.g., gas pump attendants, garage workers, printing workers, construction workers), compared to an age-matched reference group (n = 100). Serum levels of alanine aminotransferase (ALT), aspartate aminotransferase (AST), and γ-glutamyl transferase (γ-GT) were not different in the two groups. The mean lead concentrations were 78 μg/dL (SD 43) in the lead workers and 20 μg/dL (SD 12) in the reference group. Hsiao et al. (2001) found no association between blood lead concentration and ALT activity, in a longitudinal study of a group of battery manufactory workers (n = 30). Mean blood lead concentrations ranged from 60 μg/dL (approximate range 25–100 μg/dL) at the start of the study (1989) and 30 μg/dL (approximate range 10–60 μg/dL) in the final year of the study (1999).

6.9.4.3 Hepatic Cytochrome P-450 Function

Urinary excretion of 6-β-hydroxycortisol (6-β-OH-cortisol) derives primarily from oxidation of cortisol through the hepatic cytochrome P450 phenotype CYP3A4. A lower urinary 6-β-OH-cortisol:cortisol ratio is indicative of possible suppression of hepatic CYP3A4 activity. Saenger et al. (1984) found significantly lower (~45% lower) urinary excretion of 6-β-OH-cortisol and lower urinary 6-β-OH-cortisol:cortisol ratio in 2–9 year-old children (n = 26) who qualified for chelation (EDTA-provoked urinary lead >500 μg/24 h) than in children who did not qualify, and significantly lower than in an age-matched reference group. Urinary 6-β-OH-cortisol:cortisol ratio was significantly correlated with blood lead ($r = -0.514$, $p < 0.001$), urinary lead, and EDTA-provoked urinary lead ($r = -0.593$, $p < 0.001$). Mean blood lead concentrations were 46 μg/dL (range 33–60 μg/dL), prior to chelation, and 42 μg/dL (range 32-60 μg/dL) in the children who did not qualify for chelation.

Satarug et al. (2004) measured urinary excretion of 7-hydroxy-coumarin (7-OH-coumarin) following a single oral dose of coumarin to assess effects of cadmium and lead exposure on cytochrome P450 phenotype CYP2A6. The rationale for this approach is that 7-hydroxylation of coumarin occurs solely through the CYP2A6 pathway. Coumarin-induced urinary 7-OH-coumarin was measured in a group (n = 118) selected from the general population in Bangkok, Thailand. All subjects were nonsmokers. The study found a significant association between increasing urinary lead and decreasing covariate-adjusted urinary 7-OH-coumarin in males, but not in females. Covariates retained included age and zinc excretion. A significant association, in opposite direction, was found between urinary cadmium and urinary 7-OH-coumarin. Mean urinary lead levels (blood lead concentrations were not reported) were 1.3 μg/g creatinine (range 0.1–1.2 μg/dL) in males, and 2.4 μg/g creatinine (range 0.6–6.8 μg/dL) in females. Mean serum lead concentrations were 4 μg/L (range 1–28 μg/dL) in males and 3 μg/dL (range 1–12 μg/dL) in females. The range 1–28 μg/L serum would correspond to a blood lead concentration range of approximately 30–112 μg/dL (U.S. Environmental Protection Agency, 2003). These results are consistent with observations of depressed excretion of metabolites of the CYP2A6 substrate, phenazone, in association with overt clinical lead toxicity in lead workers (Fischbein et al., 1977; Meredith et al., 1977).

6.9.5 Effects of Lead on the Gastrointestinal System

6.9.5.1 Summary of Key Findings of the Effects of Lead on the Gastrointestinal System from the 1986 Lead AQCD

The 1986 Lead AQCD described gastrointestinal colic (abdominal pain, constipation, intestinal paralysis) as a consistent early symptom of lead poisoning in humans and noted that such symptoms may be present in association with blood lead concentrations in the range of 30-80 μg/dL. The 1986 Lead AQCD concluded that information was insufficient to establish clear concentration (i.e., blood concentration)-response relationships in the general population in association with environmental exposure. Subsequent to the 1986 AQCD several studies of prevalence of symptoms of gastrointestinal colic in lead workers have been reported that provide evidence for symptoms in association with blood lead concentrations >50–80 μg/dL (Awad el Karim et al., 1986; Holness and Nethercott, 1988; Lee et al., 2000; Matte et al., 1989).

Summaries of these studies are presented in Annex Table AX6-9.8. Similar types of studies of children have not been reported.

6.9.5.2 Gastrointestinal Colic

Lee et al. (2000) collected data on symptoms (self-reported questionnaire) in male lead workers (n = 95) who worked in secondary smelters, PVC-stabilizer manufacture facilities, or battery manufacture facilities. A logistic regression model was applied to the prevalence data for gastrointestinal symptoms (loss of appetite, constipation or diarrhea, abdominal pain). The covariate-adjusted odds ratio for symptoms, in association with blood lead concentration (≥versus <the group median, 45.7 μg/dL), was not significant (1.8, [95% CI: 0.7, 4.5]). The corresponding odds ratio for DMSA-provoked urinary lead (≥versus <260.5 μg/4 h, the group median) was also not significant (1.1, [95% CI: 0.4, 2.5]). However, the odds ratio for neuromuscular symptoms in association with DMSA-provoked urinary lead was significant (7.8, [95% CI: 2.8, 24.5]), suggesting that neuromuscular symptoms may occur in association with exposures that are insufficient to result in detectable gastrointestinal symptoms. Covariates retained in the final regression models were age, tobacco smoking, and alcohol consumption.

Three other studies have attempted to quantify associations between lead exposure and gastrointestinal symptoms in lead workers (Awad el Karim et al., 1986; Holness and Nethercott, 1988; Matte et al., 1989). Holness and Nethercott (1988) found a significantly ($p < 0.05$) higher prevalence of symptoms in a group of demolition workers (n = 119) in association with a blood lead concentration range 50–70 μg/dL (n = 87), 37% for abdominal cramps and 42% for constipation, or >70 μg/dL (n = 19) 77% for abdominal cramps and 62% for constipation compared to a group of workers in which the blood lead concentration range was <50 μg/dL (n = 13), prevalences of 8% and 6%. Awad el Karim et al. (1986) found higher prevalence of gastrointestinal symptoms, for abdominal colic and constipation, respectively, in male battery manufacture workers, 41.3% for abdominal colic and 41.4% for constipation, compared to a reference group of workers, n = 40 prevalences of 7.5% and 10% for abdominal colic and constipation, respectively. The blood lead ranges were 55–81 μg/dL in the lead workers and 7-33 μg/dL in the reference group. Matte et al. (1989) did not find a significant difference in prevalence of gastrointestinal symptoms (decreased appetite, nausea, abdominal pain) among a group of battery manufacture and repair workers (n = 63) when stratified by blood lead

concentration (60 µg/dL, ≥60 µg/dL). The prevalence ratio (high/low blood lead strata) for abdominal pain was 1.5 (95% CI: 0.5, 4.6).

In a small study of environmentally-exposed adults, Bercovitz and Laufer (1991) found that the lead level in the dentine of patients with gastrointestinal ulcers (n = 11), even long after recovery, were significantly higher (mean lead 75.02 µg/g [SE 8.15]) than that in healthy subjects (mean lead 25.62 µg/g [SE 10.15]). Ten of the 11 peptic ulcer patients had a higher lead level than the healthy subjects. In these 10 patients, increased severity of the ulcer and longevity of suffering was associated with increased tooth lead levels. The authors suggested that increased absorption of lead was associated with damage to the epithelial mucosal cells of the gastrointestinal tract.

6.9.6 Effects of Lead on the Respiratory System

6.9.6.1 Summary of Key Findings of the Effects of Lead on the Respiratory System from the 1986 Lead AQCD

The 1986 Lead AQCD did not discuss effects of lead on the respiratory tract on humans. Only one study since the 1986 document has examined the association between lead and respiratory health outcomes.

6.9.6.2 Pulmonary Function

Bagci et al. (2004) conducted pulmonary function tests on a group of male battery manufacture workers (n = 22), automobile exhaust repair workers (n = 40), and a group of hospital workers (n = 24). Mean blood lead concentrations were 37 µg/dL (SD 8) in the battery manufacture group, 27 µg/dL (SD 9) in the exhaust repair group, and 15 µg/dL (SD 3) in the hospital workers. Lead workers and the reference group had similar tobacco smoking prevalences (51–56%). Battery manufacture workers had significantly lower forced expiratory volume in one second (FEV_1), FEV_1:vital capacity (VC) ratio, FEV_1/forced vital capacity (FVC) ratio, forced expiration flow (FEF), and maximum voluntary ventilation (MVV) compared to the hospital workers. Blood lead concentration was significantly negatively correlated with FEV_1/FVC ($r = -0.31$, $p = 0.006$) and FEF ($r = -0.30$, $p = 0.009$) after adjusting for age, cigarette smoking, and exposure duration. Results from this study are further summarized in Annex Table AX6-9.9.

6.9.7 Effects of Lead on Bone and Teeth

6.9.7.1 Summary of Key Findings of the Effects of Lead on Bone and Teeth from the 1986 Lead AQCD

The 1986 Lead AQCD did not discuss the effects of lead on bone and teeth. Since completion of the 1986 AQCD, an additional development in lead epidemiology has been studies that have explored possible associations between lead exposure and risk of dental caries (Campbell et al., 2000; Dye et al., 2002; Gemmel et al., 2002; Moss et al., 1999). In addition, a limited number of studies also examined the toxic effect of lead on bone. These studies are summarized in Annex Table AX6-9.10.

6.9.7.2 Bone Toxicity

The number of papers dealing with direct toxicity of lead on bone is limited. Most papers are reviews (Hu et al., 1991; Puzas, 2000; Puzas et al., 1992; Rabinowitz, 1991; Silbergeld, 1991; Silbergeld et al., 1993; Vig and Hu, 2000) or based on cellular studies (e.g., Pounds et al., 1991) or animals.

Various authors have suggested that lead is a potential risk factor for osteoporosis because of the pivotal role of the skeleton in lead toxicokinetics (Goyer et al., 1994). Bone cells accumulate lead actively and earlier ideas suggested that lead was incorporated into the mineral matrix of the bone (Wittmers et al., 1988). However, in an in vivo iliac bone biopsy using laser microbeam mass analysis on a lead-intoxicated adult female following chelation therapy, Flood et al. (1988) found the extracellular lead was concentrated in the superficial 3 to 6 μm of the osteoid zone of bony trabeculae. As lead was absent from the deeper parts of the mineralized matrix, the authors suggested that lead binds more strongly to the organic matrix than to bone mineral.

There is increasing evidence from cell culture experiments, animal studies, and from measurements in humans that lead may exert detrimental effects on bone mineral metabolism. In humans this evidence comes from several studies. Following on from the earlier observations of Rosen et al. (1980) that 1,25 $(OH)_2$ vitamin D levels are reduced in lead poisoned children, Markowitz et al. (1988) found that osteocalcin levels were inversely related to lead body burden in moderately lead poisoned children. During chelation treatment for lead, the osteocalcin levels were shown to increase.

An inverse relationship between blood lead and stature and chest circumference has been observed in children from the NHANES II study (Schwartz et al., 1986). There are several explanations for the inverse correlation between blood lead and growth in children. First, blood lead level may be a composite factor for genetic, ethnic, nutritional, environmental, and sociocultural factors. Second, nutritional deficits that retard growth also enhance lead absorption. Finally, there may be a direct effect of low level lead on growth in children. This condition was explained by Dowd et al. (1994) as resulting from the inhibition by Pb^{2+} of binding of osteocalcin to hydroxyapatite. Effects similar to those described by Schwartz et al. (1986) were reported by Angle and Kuntzelman (1989), Lauwers et al. (1986), and Shukla et al. (1989).

Puzas et al. (1992) suggested lead could upset the very sensitive interactive metabolic activity of osteoblasts and chondrocytes and thereby affect bone growth. In a later review, Puzas (2000) enlarged upon his earlier paper and described in more detail the potential mechanism of lead on growth plate cartilage metabolism and effects of lead on osteoclasts and osteoblasts, especially associated with osteoporosis.

Observational studies by Spencer et al. (1992, 1994) suggested a link between occupational exposure to lead and Paget's disease in both males and females but the authors declined to advocate a causal effect. Later Spencer et al. (1995) found that 92% of a group of 48 patients with Paget's disease were exposed to lead either from occupational or environmental sources. Adachi et al. (1998) explored a possible association between lead and bone disease from XRF analyses of cortical and trabecular bone lead content in 117 patients who attended a metabolic bone disease clinic (n = 92) or were undergoing dialysis for renal failure (n = 25). In patients suffering from Paget's disease, cortical bone lead content was higher than it was in controls, patients with osteoporosis, and patients on dialysis. Trabecular bone lead content was lowest in patients with Paget's disease or osteitis fibrosa. However, the authors could not distinguish between two alternatives, the first being that increased bone turnover due to Paget's disease releases lead from trabecular bone that is then available for deposition into cortical bone, or secondly, that an increased lead content in cortical bone may cause increased turnover with release of lead from trabecular bone.

In another facet of the Normative Aging Study, Shadick et al. (2000) investigated a possible association between long-term lead accumulation and hyperuricemia and gouty arthritis

in 777 male subjects. They found a positive association between patella bone lead and uric acid levels ($p = 0.022$) but no association between bone or blood lead and gout in this environmentally-exposed group.

6.9.7.3 Dental Health

Caries is considered an infectious disease arising from a multifactorial process involving particular flora, dietary exposures, and a susceptible host (Schafer and Adair, 2000). Increased caries risk has been detected in association with increasing blood lead concentrations in populations whose mean blood lead concentrations are approximately 2–3 µg/dL (Dye et al., 2002; Gemmel et al., 2002; Moss et al., 1999).

Several studies have examined relationships between lead exposure and the occurrence of dental caries in children and adults. The two largest studies were analyses of data collected in the NHANES III; both found significant associations between increasing caries prevalence and increasing blood lead concentrations in children and adolescent (Moss et al., 1999) and the adult (Dye et al., 2002) populations, whose geometric mean blood lead concentration was ~2.5 µg/dL. In the Moss et al. (1999) study, the odds ratios for caries in association with a 5 µg/dL increase in blood lead concentration (i.e., from <2 µg/dL) was 1.8 (95% CI: 1.3, 2.5). Outcomes of two smaller studies were mixed, with one study finding no significant association between blood lead and caries prevalence (Campbell et al., 2000) and one study finding significant associations (Gemmel et al., 2002); the latter, in children whose mean blood lead concentration was 2.9 µg/dL (maximum 13 µg/dL).

The Moss et al. (1999) NHANES III analysis included the results of coronal caries examinations on 24,901 subjects, stratified by age: 2–5 years (n = 3,547), 6–11 years (n = 2,894), and ≥12 years (n = 18,460). Specific outcomes assessed varied by age group: for children 2–11 years who had at least one deciduous tooth, the number of deciduous teeth displaying decayed or filled surfaces (DFS); for subjects ≥6 years and who had at least one permanent tooth, the number of permanent teeth displaying decayed or filled surfaces; and for subjects ≥12 years, the sum of decayed, missing, and filled surfaces on permanent teeth (DMFS). In a multivariate linear regression model, increasing blood lead concentration (log-transformed) was significantly associated with covariate-adjusted increases in dfs in the 2–5 year age group ($\beta = 1.78$ [SE 0.59], $p = 0.004$) and in the 6–11 year age group ($\beta = 1.42$ [SE 0.51], $p = 0.007$). Log-transformed

blood lead also was associated with increases in DFS in the 6-11 years age group (β = 0.48 [SE 0.22], p = 0.03) and in the ≥12 years age group (β = 2.50 [SE 0.69], p<0.001), and increases in DMFS in the ≥12 years age group (β = 5.48 [SE 1.44], p = 0.01). The odds ratios (compared to 1st tertile, ≤1.66 μg/dL) for the binomial outcome, 0 or ≥1 DMFS, were 1.36 (95% CI: 1.01, 2.83) for the blood lead concentration range 1.66-3.52 μg/dL, and 1.66 (95% CI: 1.12, 2.48) for the range >3.52 μg/dL. Corresponding population risks attributable to blood lead concentration were 9.6% and 13.5% in the blood lead strata, respectively. An increase in blood lead of 5 μg/dL was associated with an odds ratio of 1.8 (95% CI: 1.3, 2.5). Covariates included in the models were age, gender, race/ethnicity, poverty income ratio, exposure to cigarette smoke, geographic region, educational level of head of household, carbohydrate and calcium intakes, and frequency of dental visits.

Gemmel et al. (2002) conducted a cross-sectional study of associations between blood lead concentration and dental caries in children, 6-10 years of age (n = 543), who resided either in an urban (n = 290) or rural (n = 253) setting. Mean blood lead concentrations were 2.9 μg/dL (SD 2.0, maximum 13 μg/dL) in the urban group and 1.7 μg/dL (SD 1.0, maximum 7 μg/dL) in the rural group. Increasing blood lead concentration (ln-transformed) was significantly associated with covariate-adjusted number of caries (dfs + DFS) (ln-transformed) in the urban group (β = 0.22 [SE 0.08], p = 0.005), but not in the rural group (β = –0.15 [SE 0.09], p = 0.09). When dfs counts were stratified by permanent or deciduous teeth, the blood lead association in the urban group was significant for deciduous teeth (β = 0.28 [SE 0.09], p = 0.002), but not for permanent teeth (β = 0.02 [SE 0.07], p = 0.8). Covariates retained in the linear regression model were age, sex, ethnicity, family income, education of female guardian, maternal smoking, frequency of tooth brushing, firmness of toothbrush bristles, and frequency of chewing gum.

Campbell et al. (2000) was a retrospective cohort study in which dfs were assessed in children 7-12 years of age (n = 248) from Rochester, NY. Mean blood lead concentration, measured at ages 18 and 37 months of age, was 10.7 μg/dL (range 18.0-36.8 μg/dL). The covariate-adjusted odds ratios for caries associated with a blood lead concentration >10 μg/dL compared to ≤10 μg/dL were 0.95 μg/dL (95% CI: 0.43, 2.09) for permanent teeth and 1.77 μg/dL (95% CI: 0.97, 3.24) for deciduous teeth. Covariates retained in the logistic model were age, grade in school, number of tooth surfaces at risk. Other covariates examined in the models, all of which had no significant effect on the outcome, were gender, race/ethnicity, SES,

parental education, residence in community supplied with fluoridated drinking water, and various dental hygiene variables. This study did not demonstrate that lead exposure >10 µg/dL as a toddler was a strong predictor of caries among school-age children, but the authors noted that this might be due to limited statistical power.

Dye et al. (2002) analyzed data collected in NHANES III on indices of periodontal bone loss. The analysis was confined to subjects 20-69 years of age (n = 10,033). The geometric mean blood lead concentration of the study group was 2.5 µg/dL (SE 0.08), with 2.4% of the group having blood lead levels >10 µg/dL. Increasing log-transformed blood lead was significantly associated with increasing prevalence of covariate-adjusted dental furcation (β = 0.13 [SE 0.05], p = 0.005). Dental furcation is indicative of severe periodontal disease. Covariates retained in the linear regression model were age, sex, race/ethnicity, education, smoking, and age of home. Smoking status was a significant interaction term when included in the model (β = 0.10 [SE 0.05], p=0.034). When stratified by smoking status, the association between dental furcation and blood lead concentration was significant for current smokers (β = 0.21 [SE 0.07], p = 0.004) and former smokers (β = 0.17 [SE 0.07], p=0.015), but not for nonsmokers (β = –0.02 [SE 0.07], p = 0.747).

Some studies examined the relationship between tooth lead concentrations and dental caries. In their compilation of metal concentrations in 1,200 deciduous teeth from a Norwegian population, Tvinnereim et al. (2000) found that carious teeth had higher lead concentrations than noncarious teeth. Gil et al. (1994) measured lead concentrations from 220 whole deciduous and permanent teeth from Coruna, Spain. The geometric mean lead level was 10.36 µg/g of tooth. There was a significant increase in teeth lead levels with advancing age. Permanent teeth showed higher mean lead values (13.09 µg/g [SEM 1.07]) than deciduous teeth (3.96 µg/g [SEM 1.07]). The authors reported a possible relationship between increased lead content and periodontal pathology but did not observe any relationship between caries and lead concentrations.

6.9.8 Effects of Lead on Ocular Health

6.9.8.1 Summary of Key Findings of the Effects of Lead on Ocular Health from the 1986 Lead AQCD

The 1986 Lead AQCD did not address effects of lead on ocular health in humans. Various disturbances of the visual system have been observed in association with overt clinical lead poisoning, including retinal stippling and edema, cataracts, ocular muscle paralysis, and impaired vision (see Otto and Fox, 1993 for review). Two longitudinal studies completed since 1986 provide evidence for possible associations between lead exposure and visual evoked retinal responses in children of mothers whose blood lead concentrations in mid-pregnancy were in the range of 10–32 μg/dL (Rothenberg et al., 2002), and evidence for a possible association between lead exposure and risk of cataracts in middle-aged males whose tibia bone lead levels were in the range 31–126 μg/g (Schaumberg et al., 2004). These studies are summarized in Annex Table AX6-9.11.

6.9.8.2 Ocular Effects

In the Mexico City prospective lead study, Rothenberg et al. (2002) measured flash-evoked electroretinograms (ERG) in a subset of the study group (n = 45) at ages 7–10 years. As part of the prospective study, blood lead concentrations had been measured during pregnancy and in the children, at birth and every 6 months, thereafter. Increasing maternal blood lead, measured at 12 weeks of gestation, was significantly associated with increasing ERG a-wave and b-wave amplitude, with significant increases in a-wave in the second maternal blood lead tertile (range 6.0–10.0 μg/dL), and a-wave and b-wave in the third maternal blood lead tertile (range 10.5–32.5 μg/dL), compared to the first blood lead tertile (range 2.0–5.5 μg/dL). No other blood lead measurements were significantly associated with any ERG outcomes.

As part of the longitudinal Normative Aging Study, Schaumberg et al. (2004) analyzed prevalence of cataracts in adult males (n = 642), mean age 69 years (range 60–93). Subjects were stratified by blood lead, patella bone lead, or tibia bone lead quintiles for a logistic regression analysis of the odds ratios for cataracts (first quintile as reference). Covariate adjusted odds ratio for cataracts in the fifth tibia bone lead quintile was significant (3.19 [95% CI: 1.48, .90]). Odds ratios for cataracts were not significantly associated with patella bone lead (1.88 [95% CI: 0.88, 4.02]) or blood lead (0.89 [95% CI: 0.46, 1.72]). The first and fifth

quintile lead levels were 0–11 μg/g and 31–126 μg/g for tibia bone; 1–16 μg/g and 43–165 μg/g for patella bone; and 1.0–3.0 μg/g and 8–35 μg/g for blood. Covariates retained in the regression model were age, smoking, history of diabetes; and daily intake of vitamin C, vitamin E, and carotenoids.

Cavalleri et al. (1982) measured visual fields of male workers in a polyvinyl pipe manufacturing facility (n = 35) who were exposed to lead stearate. Workers in a reference group (n = 350) were individually matched for age, smoking, and alcohol consumption. Visual sensitivity was significantly lower in lead workers compared to the reference group; however, visual sensitivity index was not significantly associated with blood or urine lead. Prevalence of scotoma in the mesopic field was 28.5% in the lead workers and 0% in the reference group. Mean blood lead levels were 46 μg/dL (range 21–82 μg/dL) in the lead workers and 30 μg/dL (range 21-42 μg/dL) in the reference group.

6.9.9 Summary of the Epidemiologic Evidence for the Effects of Lead on Other Organ Systems

Biochemical Effects of Lead

Evidence for disruption of heme synthesis derives from numerous studies in which lead exposure has been associated with decreased activities of enzymes in the heme synthesis pathway (i.e., ALAS, ferrochelatase) and increased levels of substrates for heme synthesis (i.e., ALA, coproporphyrin, erythrocyte protoporphrin) in both children and adults. Quantitative relationships between blood lead concentration and the above biomarkers of impaired heme synthesis are highly consistent across studies (e.g., Alessio et al., 1977, 1976; Gennart et al., 1992; Hernberg et al., 1970; Morita et al., 1997; Oishi et al., 1996; Piomelli et al., 1982; Roels and Lauwerys, 1987; Selander and Cramer, 1970; Soldin et al., 2003; Wildt et al., 1987). Increases in blood lead concentration of approximately 20–30 μg/dL are sufficient to halve erythrocyte ALAD activity and sufficiently inhibit ferrochelatase to double erythrocyte protoporphyrin levels.

Associations between occupational exposure to lead and changes in blood lipid composition have been observed. These include increased levels of lipid peroxides in blood and/or serum (Jiun and Hsien, 1994; Sugawara et al., 1991; Ito et al., 1985) and increased serum levels of total and HDL cholesterol (Kristal-Boneh et al., 1999). Effects on serum cholesterol

levels were evident in association with a mean blood lead concentration of 42 µg/dL (Kristal-Boneh et al., 1999) or a range of 5–62 µg/dL (approximated mean 14 µg/dL) (Ito et al., 1985). Oxidative changes in blood lipids (e.g., increased levels of lipid peroxides and malondialdehyde levels) as well as decreased levels of erythrocyte superoxide dismutase, catalase, G6PD, and GSH peroxidase; and increased lymphocyte reactive oxygen species and depleted GSH levels, indicative of increased oxidative stress, have been observed in lead workers in association with blood lead concentrations >30 µg/dL (Fracasso et al., 2002; Ito et al., 1985; Jiun and Hsien, 1994; Solliway et al., 1996; Sugawara et al., 1991).

Disruption of Hemoglobin Synthesis and Declines in Erythrocyte Numbers

Exposures that result in blood lead concentrations below 40 µg/dL appear to be tolerated without a decline in blood hemoglobin levels or hematocrit. However, perturbation of erythropoiesis, indicated by changes in serum erythropoietin and progenitor cells, occurs in association with blood lead concentrations below 40 µg/dL and in the absence of detectable changes in blood hemoglobin levels or hematocrit in children (Graziano et al., 2004; Liebelt et al., 1999) and adults (Graziano et al., 1990; Osterode et al., 1999; Romeo et al., 1996). Risk of clinical anemia in children becomes appreciable at much higher blood lead concentrations; a 10% decrease in hematocrit has been estimated to occur in association with blood lead concentrations ≥85 µg/dL; a 10% probability of anemia (hematocrit <35%) was estimated to be associated with a blood lead concentration of approximately 20 µg/dL at age 1 year, 50 µg/dL at age 3 years, and 75 µg/dL at age 5 years. (Schwartz et al., 1990). In adults, with blood lead levels below 25 µg/dL, increasing patella bone lead, but not blood lead, was associated with a significant decrease in hematocrit.

Effects on the Endocrine System

Several studies have examined possible associations between lead exposures in children and adults and various biomarkers of endocrine function, including the thyroid, male reproductive, and calcitropic endocrine systems. The strongest study designs have yielded no associations, or weak associations, between lead exposure and thyroid hormone status (Erfurth et al., 2001; Schumacher et al., 1998; Tuppurainen et al., 1988; Zheng et al., 2001). Studies of occupational exposures which included subjects having blood lead concentrations exceeding

100 µg/dL have found depression of serum T3 and/or T4 levels, without a detectable increase in serum TSH; however, studies in which the blood lead distribution was dominated by levels well below 100 µg/dL, have found either no effects or subclinical increases in serum T3, T4, with no change in TSH levels.

Studies of the male reproductive system that attempted to control for confounding effects of age have yielded mixed outcomes (Alexander et al., 1998, 1996; Erfurth et al., 2001; Gustafson et al., 1989; McGregor and Mason, 1990; Ng et al., 1991). Blood lead ranges in these studies were similar (4–90 g/dL), yet outcomes were mixed, with no change (Erfurth et al., 2001; Gustafson et al., 1989; McGregor and Mason, 1990), or subclinical decrease (Alexander et al., 1998, 1996; Ng et al., 1991) in serum testosterone (TES) in association with lead exposure. There are also mixed effects on serum follicle stimulating hormone (FSH) and luteinizing hormone (LH) with increases (McGregor and Mason, 1990; Ng et al., 1991), decreases (Gustafson et al., 1989), and with no change (Alexander et al., 1998, 1996; Erfurth et al., 2001) in hormone levels observed. The inconsistency in the direction of effects on TES and the two androgen-regulating pituitary hormones, FSH and LH, is particularly noteworthy, in the absence of evidence for effects of lead exposure on GNRH-induced FSH (Erfurth et al. 2001).

Children exposed to relatively a high level of lead >30 µg/dL may exhibit depressed levels of circulating 1,25-OH-D (Mahaffey et al., 1982; Rosen et al., 1980). However, associations between serum vitamin D status and blood lead may not be present in calcium-replete children who have average lifetime blood lead concentrations below 25 µg/dL (Koo et al., 1991). In adults, exposures to lead that result in blood lead concentrations >40–60 µg/dL may increase, rather than decrease, circulating levels of 1,25-OH-D and PTH (Kristal-Boneh et al., 1999; Mason et al., 1990).

Effects on the Hepatic System

Few studies of hepatic effects of lead on humans have been reported since the 1986 Lead AQCD. Studies of hepatic enzyme levels in serum suggest that liver injury may be present in lead workers; however, associations specifically with lead exposures are not evident (Al-Neamy et al., 2001; Hsiao et al., 2001). Studies of urinary metabolites of cytochrome P450 phenotypes CYP2A6 and CYP3A4 suggest possible associations between lead exposure and suppression of hepatic enzyme activity. The effect on CYP2A6 activity was observed in children with high lead

burdens (i.e., blood lead concentration >40 μg/dL, EDTA-provoked urinary lead >500 μg/dL). The effect on CYP3A4 was observed in association with blood lead ranges of approximately 30-112 μg/dL (based on reported serum lead concentrations).

Effects on the Gastrointestinal System

Several studies of prevalence of symptoms of gastrointestinal colic in lead workers provide evidence for symptoms in association with blood lead concentrations >50–80 μg/dL (Awad el Karim et al., 1986; Holness and Nethercott, 1988; Lee et al., 2000; Matte et al., 1989). Similar types of studies of children have not been reported.

Effect on Bone and Teeth

There is limited, but suggestive evidence of an association between lead exposure and bone toxicity. However, in most studies, it is difficult to assess the direct contribution of lead on bone diseases or reduced growth. Several studies that have explored possible associations between lead exposure and risk of dental caries (Campbell et al., 2000; Dye et al., 2002; Gemmel et al., 2002; Moss et al., 1999). Increased caries risk has been detected in association with increasing blood lead concentrations in populations whose mean blood lead concentrations are approximately 2-3 μg/dL (Dye et al., 2002; Gemmel et al., 2002; Moss et al., 1999).

Ocular Health

Various disturbances of the visual system have been observed in association with overt clinical lead poisoning, including retinal stippling and edema, cataracts, ocular muscle paralysis, and impaired vision (Otto and Fox, 1993). Two longitudinal studies completed since the 1986 Lead AQCD provide evidence for possible associations (a) between lead exposure and visual evoked retinal responses in children of mothers whose blood lead concentrations in mid-pregnancy was 10.5–32.5 μg/dL (Rothenberg et al., 2002) and (b) between lead exposure and risk of cataracts in middle-aged males whose tibia bone lead levels were 31-126 μg/g (Schaumberg et al., 2004).

6.10 INTERPRETIVE ASSESSMENT OF THE EVIDENCE IN EPIDEMIOLOGIC STUDIES OF LEAD HEALTH EFFECTS

6.10.1 Introduction

A remarkable expansion has occurred since the 1990 Lead Supplement in the extent of the database available for drawing inferences about the various expressions of lead toxicity. Moreover, the nature of the evidence available has changed as well. Many of the studies conducted prior to 1990 focused on the issue of whether an observed observation was likely to be real or the result of chance, selection bias, residual confounding, or some other methodological error. The validity of any association still needs to be assured. The studies since 1990 mainly focus on characteristics of the pertinent concentration-response relationships, including the functional forms of the relationships, the slopes of the relationships, the natural histories of adverse effects, and the effect modifying influences of various co-exposures and host characteristics.

6.10.2 Exposure and Outcome Assessment in Lead Epidemiologic Studies

6.10.2.1 Assessment of Lead Exposure and Body Burdens Using Biomarkers

For any health endpoint of interest, the most useful biomarker of exposure is one that provides information about the lead dose at the critical target organ and, moreover, reflects the exposure averaging time that is appropriate to the underlying pathogenetic processes (e.g., cumulative over lifetime, cumulative over a circumscribed age range, concurrent, etc.). In recent studies of lead and health, the exposure biomarkers most frequently used are blood lead and bone lead. For outcomes other than those relating to hematopoeisis and bone health, these biomarkers provide information about lead dose that is some distance from the target organ. For example, given that the central nervous system is considered the critical target organ for childhood lead toxicity, it would be most helpful to be able to measure, in vivo, the concentration of lead at the cellular site(s) of action in the brain. Because such measurements are not currently feasible, however, investigators must rely on measurements of lead in the more readily accessible but peripheral tissues. The relationship between brain lead and lead in each of these surrogate tissues is still poorly understood, although the pharmacokinetics clearly differs among these compartments. In both rodents and nonhuman primates, brain lead level falls much more slowly than blood lead level following chelation with succimer and, in the rodent, in nonchelated

animals after cessation of exposure. These observations suggest that using blood lead as an index of lead in the brain will result in exposure misclassification, although the magnitude of this bias in any specific setting will be difficult to characterize. The most likely direction, however, would be underestimation of the amount of lead in the brain, at least under scenarios involving chronic exposure.

As an exposure biomarker, blood lead level has other limitations. Only about 5% of an individual's total body lead burden resides in blood. Furthermore, blood consists of several sub-compartments. More than 90% of lead in whole blood is bound to red cell proteins such as hemoglobin, with the balance in plasma. From a toxicological perspective, this unbound fraction is likely to be the most important sub-compartment of blood lead because of the ease with which it diffuses into soft tissues. The concentration of lead in plasma is much lower than in whole blood, however. For example, in a group of pregnant women with blood lead levels below 10 μg/dL, plasma lead levels were less than 0.3% of the whole blood lead level. The greater relative abundance of lead in whole blood makes its measurement much easier (and more affordable) than the measurement of lead in plasma. The use of whole blood lead as a surrogate for plasma lead could be justified if the ratio of whole blood lead to plasma lead were well characterized, but this is not so. At least some studies suggest that it varies several-fold among individuals with the same blood lead level. Moreover, the ability of red cells to bind lead is limited, so the ratio of blood lead to plasma lead would be expected to be nonlinear. Thus, interpreting whole blood lead level as a proxy for plasma lead level, which, itself, is a proxy for brain lead level, will result in some exposure misclassification.

Another limitation in the use of blood lead as the exposure biomarker is that its residence time in blood is closely linked to red cell lifetime, with a half-time on the order of 30 days. Thus, a high blood lead level does not necessarily indicate a high body lead burden. Similarly, individuals who have the same blood lead level will not necessarily have similar body burdens or exposure histories. The rate at which blood lead level changes with time/age depends on exposure history due to re-equilibration of lead stored in the various body pools. In nonchelated children, the time for blood lead to decline to a value less than 10 μg/dL was linearly related to baseline blood lead level. A single blood lead measurement might therefore provide limited information about an individual's lead exposure history, a difficulty frequently cited with respect to the interpretation of cross-sectional studies of pediatric lead toxicity, in which children's blood

lead level is often measured only once, and sometimes only well after the period when levels typically peak (18-30 months). If it is exposures to lead in the early postnatal years that are most detrimental to children's development, categorizing a child's exposure status based on the blood lead level that is contemporaneous with the measurement of neurodevelopment at school-age could result in exposure misclassification. Unless intra-individual stability of serial blood lead levels is very high within a study cohort, misclassification would probably be non-differential, more likely resulting in an underestimate rather than an overestimate of the effect of lead on child neurodevelopment (Jurek et al., 2005). This concern must be qualified, however, by recent data from some longitudinal studies indicating that concurrent blood lead level, even at ages well beyond 18 to 30 months, is sometimes the strongest predictor of late outcomes (Dietrich et al., 1993a,b; Canfield et al., 2003a; Tong et al., 1996; Wasserman et al., 2000b). Age-related changes in vulnerability, and the reasons why it might differ across studies, remain uncertain. It might be that among children with chronically elevated exposure, but not in children with relatively low lifetime exposure, blood lead level measured at school-age is a reasonably good marker of cumulative exposure. That concurrent blood lead level is, under some circumstances, a stronger predictor of school-age outcomes than is blood lead level in the early postnatal years does not necessarily imply greater vulnerability of the brain to ongoing than to past exposure.

The development of X-ray-fluorescence (XRF) methods for measuring lead in mineralized tissues offers another approach for characterization and reconstruction of exposure history. Such tissues are long-term lead storage sites, with a half-life measured in decades and contain approximately 90% of the total body lead burden in adults and 70% in children. Thus, bone lead is an index with a long exposure averaging time. XRF methods have proven useful in studying individuals with occupational lead exposure, those living in highly polluted environments, and those for whom community lead exposures are or, in the past, were relatively high (e.g., Korrick et al., 1999; Schwartz et al., 2000a,b,c,d). In a relatively highly exposed cohort of pregnant women in Mexico City, higher bone lead levels at one month postpartum were associated with reduced birth weight, less infant weight gain, smaller head circumference and birth length, and slower infant development (Gomaa et al., 2002; Gonzalez-Cossio et al., 1997; Hernandez-Avila et al., 2002; Sanin et al., 2001). Among children living near a large lead smelter in Yugoslavia, IQ at age 10-12 years was more strongly associated, inversely, with tibia lead level than with blood lead level (Wasserman et al., 2003).

Current XRF methods for measuring bone lead levels have limitations, however. Temporal features of exposure history cannot readily be discerned. Some progress has been made toward this goal by examining the spatial distribution of lead in teeth in relation to the relative abundance of stable lead isotopes, but the specialized technologies needed to carry out these analyses are unlikely ever to be widely available, and the unpredictability of tooth exfoliation makes this tissue difficult to collect unless the study design involves contact with (and the cooperation of) participants at the appropriate ages. Current XRF methods might not be sufficiently sensitive for studies of the health effects of low-dose community exposures. The bone lead levels of a large percentage of subjects might be below the detection limit, e.g., 80% in a case-control study of bone lead levels and juvenile delinquency in which the minimum detection limit was 21.5 μg/g bone mineral (Needleman et al., 2002). Even among individuals known to have histories of substantial lead exposures, such as adolescents and young adults who grew up near the Bunker Hill smelter in Idaho (McNeill et al., 2000), bone lead levels tend to be low. Lead appears to be deposited at sites of most active calcification. In children, this is trabecular bone, in which the rate of fractional resorption in early childhood is high. Depending on the amount of the child's ongoing exposure, lead deposited in bone might not remain there for decades, making bone lead level an imprecise index of lifetime lead exposure. This concern also exists in the use of tooth lead to represent cumulative lead exposure in children. Rabinowitz et al. (1993) observed that a child's tooth lead level was more strongly related to blood lead level around the time of tooth exfoliation than to an integrated index of blood lead level prior to exfoliation. Finally, it is difficult to compare the performance of different laboratories using XRF methods to measure bone lead because of the absence of standard reference materials. Nevertheless, efforts continue to modify the instrumentation or measurement protocols to reduce the detection limit.

A major research need is the development and validation of biomarkers of critical dose that, compared to blood lead or bone lead, are fewer toxicokinetic steps removed from the sites of lead's actions in the brain. One promising front in the effort to deduce the contents of the "black box" separating external dose and clinical disease is the measurement of processes and products that potentially mediate the association between them. For example, magnetic resonance spectroscopy (MRS) has been used in small case series to measure the ratio of N-acetylaspartate (NAA) to creatine, which are a marker of neuronal and axonal damage and

thus, an early biological effect rather than a biomarker of exposure. In children, higher lead exposures are associated with lower NAA to creatine ratios in the frontal gray matter and, to a lesser extent, in frontal white matter (Trope et al., 1998, 2001). Similarly, an adult who had higher bone and blood lead levels than did his monozygotic twin had both greater neuropsychological deficits and lower NAA to creatine ratios in the hippocampus, frontal lobe, and midbrain (Weisskopf et al., 2004). While much remains uncertain about the interpretation of MRS, the use of this and other biochemical imaging methods, in combination with more conventional structural and functional imaging methods, might bring us closer to understanding the mechanisms of lead neurotoxicity. With the number of toxicokinetic steps separating lead levels at the critical target organs from the usual exposure biomarkers, the progress made in characterizing the concentration-response relationships is remarkable.

6.10.2.2 Assessment of Health Outcomes

Outcome measurement and outcome classification have generally received less attention from investigators than have exposure measurement and misclassification. The specific problems are, to some extent, endpoint domain-specific. With regard to neurodevelopmental toxicities, critical issues are whether the assessment instruments used are psychometrically sound and appropriate for the study cohort, the data generated will support adequate tests of the study hypotheses, and whether the instruments have been administered and scored consistently and correctly. With regard to the cardiovascular toxicities of increased blood pressure/prevalence of hypertension, the critical issue is whether the blood pressure value recorded for a participant is an accurate estimate. Multiple measurements of blood pressure are frequently made in a study but investigators usually have not taken advantage of the collected information to quantify the amount of error in the measurements. This information can be used to improve the reliability of the measurements, which would be expected to improve the precision of the associations estimated. Similarly, aggregating scores to estimate latent variables representing, for instance, "language skills" or "visual-spatial skills" is an approach that might take advantage of the overlapping information provided by the multiple tests included in neurobehavioral test batteries, producing more reliable endpoint variables. This approach, however, has not been widely applied in lead studies.

6.10.3 Concentration-Response Relationship of Lead Health Effects

Recent studies have not altered the consensus that the developing nervous system is the organ system that is most sensitive to lead toxicity in children and adults. Neurobehavioral deficits appear to occur at lower levels of exposure than do other adverse health effects, although adverse effects in other organ systems have been observed in some susceptible populations at similarly low levels (e.g., adverse renal outcomes in individuals with hypertension or chronic renal insufficiency). Effects have been reported at blood lead levels as low as 1 to 2 μg/dL in the case of neurobehavioral toxicity. Accumulating data appear to validate well the statement made in the 1996 AQCD and Addendum, and 1990 Supplement that adverse effects occur at blood lead levels of 10 to 15 μg/dL or "possibly lower." In a recent study of 6 to 16 year old children in the NHANES III survey, concentration-related deficits in reading and arithmetic scores were found even when analyses were restricted to children with concurrent blood lead levels below 5 μg/dL (Lanphear et al., 2000).

Canfield et al. (2003a) applied semi-parametric models with penalized splines to their data, essentially allowing the data to reveal the functional form that best described them. These analyses showed that the IQ decline per μg/dL increase in blood lead was greater below 10 μg/dL than it was above 10 μg/dL. The estimated slope of the IQ decline per μg/dL was greatest among children for whom the maximum blood lead level measured over the course of the study never exceeded 10 μg/dL. A similarly steeper slope at lower than at higher blood lead levels was found in a re-analysis of the Boston prospective study (Bellinger and Needleman, 2003).

Identifying the functional form that best fits a particular set of data and that presumably serves as the best description of the pertinent underlying concentration-response relationship is clearly important. The linear model (Figure 6-10.1) is, as the name implies, linear over the entire range of the exposure data. For certain tests, the assumption is made that the residuals (observed – predicted response) are normally distributed with constant variance, but violations of this assumption (heteroscedasticity) have no real effect on the estimation and minimal effect on the tests of significance (see Annex Section AX6.10). If heteroscedasticity is present but all other conditions are met, regression still yields unbiased estimators, but the standard errors can be larger than when remedial efforts such as using weighted regression are employed. The use of regression requires no assumption concerning the distribution of the independent variable

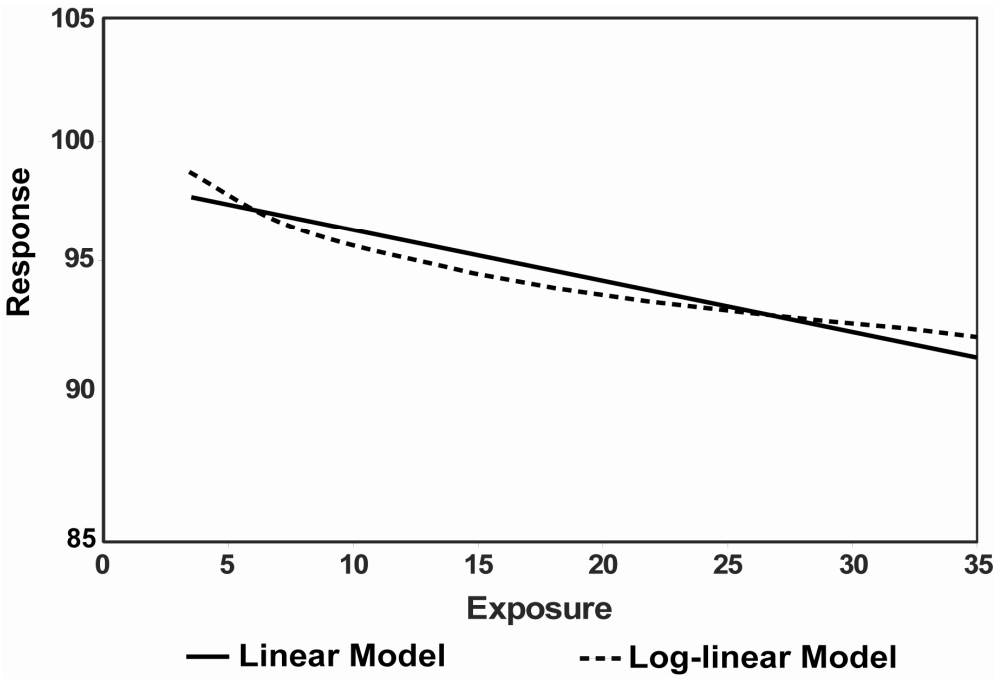

Figure 6-10.1. Comparison of a linear and log-linear model to describe the relationship between exposure and response.

(lead exposure marker). However, when the form of the heteroscedasticity is an increase in variance with level of blood lead and when the data are lognormally distributed or otherwise skewed, there are possibly a large number of influential data points at high blood lead where the data is least reliable. In this case, a log transformation of blood leads may result in more precise estimation of the slope parameter. The presence of heteroscedasticity and other departures from assumptions forming the basis for regression analysis can be detected by using diagnostic tests or graphics. These are rarely used in epidemiologic studies of lead health effects.

The log-linear model (see Figure 6-10.1) is written as:

$$\text{Response} = \alpha + \beta \, \text{Ln}(\text{lead exposure marker}),$$

where Ln is the natural logarithmic function. The log-linear model is concave upwards (assuming that the estimated coefficient is negative). It approaches a linear function for very high exposure values, but approaches infinity at very low exposure values. In other words, it is assumed that the adverse effect of lead is greater at lower than at higher blood lead levels.

Blood lead levels have been shown repeatedly to follow a lognormal distribution (Azar et al., 1975; Billick et al., 1979; Hasselblad and Nelson, 1975; Yankel et al., 1977; Hasselblad et al., 1980; U.S. Environmental Protection Agency, 1986), but this fact is not an argument for choosing the log-linear model. The choice of either log-linear or linear may be based on the Akaike's Information Criteria (Akaike, 1973), J-test (Davidson and MacKinnon, 1981), or other statistical tests if the choice is to be based on the best fitting model. Rothenberg and Rothenberg (2005) compared the linear lead model with the log-linear lead model for the pooled data from Lanphear et al. (2005) using the J-test. The J-test showed that the log lead specification was still significant ($p = 0.009$) in a model that also included the linear lead specification, indicating that the log lead specification described the data significantly better than did the linear lead specification. Other models have been used, such as nonparametric models, spline functions, and polynomial models, but the vast majority of the analyses have used either a linear model or a log-linear model.

Nonlinear concentration-response relationships are not uncommon in toxicology, although many of these are claimed to be examples of hormesis, with the lowest doses of a toxicant being associated with a beneficial effect rather than a greater adverse effect. A biological mechanism for a steeper slope at lower than at higher blood lead levels has not been identified. Perhaps the predominant mechanism at very low blood lead levels is rapidly saturated, and that a different, less rapidly saturated process becomes predominant at blood lead levels greater than 10 μg/dL. This ad hoc explanation is more descriptive than explanatory, however, and the specific processes that would produce this result have not yet been identified. Nevertheless, relationships of this apparent form have been observed in several data sets, indicating the need to determine whether such a relationship is real or a statistical artifact.

An important caveat regarding efforts to specify the functional form of the concentration-response relationship is that the accuracy that can be achieved is constrained by the extent to which the biomarker of lead concentration does, in fact, reflect the concentration at the critical target organ, the brain. The greater the misclassification, the more uncertain will be the biological relevance of the best statistical description of the concentration-response relationship.

6.10.4 Interindividual Variability in Susceptibility to Lead Toxicity

Although increased lead exposure has been linked to adverse health effects in many different organ systems, scatterplots reveal tremendous variability of observed points about the best fit lines representing the concentration-response relationships. In other words, individuals for whom the lead biomarker measured has the same value can have markedly different values on the health indicator measured. Even for neurobehavioral deficits in children, the correlation between biomarker level and test score rarely exceeds −0.2, indicating that the explained variance in the test score generally does not exceed 5%. A major challenge is therefore to decompose this variability, to distinguish components of it that reflect error from components that reflect biological processes that determine an individual's response to lead.

Deviation of the observed points from the fitted point can have many sources. Exposure misclassification is one source. The lead biomarker measured might not adequately capture the lead dose delivered to the target organ and at the time that is most appropriate biologically. In general, the error would be expected to be non-differential, i.e., it would not introduce a systematic bias in estimation of the concentration-response relationship. On average, such misclassification would be expected to result both in an attenuation of the slope of the concentration-response relationship and an increase in the scatter of the observations. As focus shifts to the risks associated with lower and lower levels of lead exposure, the importance of errors introduced by poor dosimetry will assume greater importance insofar as the effects at such levels will presumably be more subtle and increasingly difficult to detect amid the noise contributed by exposure misclassification. Outcome misclassification is another source of error that is likely to contribute to apparent interindividual variability in response. This results if the indicator of the critical health effect that is measured is fallible, i.e., an imperfect measure of the target function. Such misclassification would generally be expected to be non-differential, introducing random noise rather than a systematic bias.

Another likely source of scatter in observed points is true interindividual variability in response to a given lead dose. That is, the magnitude of individual response to lead might depend on other characteristics of that individual. Three major categories of such effect modifying factors that might influence susceptibility to lead toxicity are genetic polymorphisms, nutritional status, and social environmental factors. Adequate data are not available to provide a quantitative estimate of the amount of interindividual variability in susceptibility to lead.

6.10.4.1 Influence of Genetic Polymorphisms on Risk

Genetic polymorphisms that are presumed to influence lead toxicokinetics and/or toxicodynamics have been identified, mostly in studies of adults who were occupationally exposed to lead. Compared to workers with the wild type allele of amino levulinic acid dehydratase, workers with the variant allele had a higher mean blood lead level, greater lead-associated renal dysfunction, and an increased risk of amyotrophic lateral sclerosis (Kamel et al., 2003). Lead workers with the ATP1A2(3′) polymorphism appear to be at increased risk of lead-associated effects on blood pressure (Glenn et al., 2001). The slope of the association between floor dust lead and blood lead is steeper among children with the less common variant of the vitamin D receptor (Fox 1 or B) than among children with the wild-type allele (Haynes et al., 2003). In adults, these same alleles are associated with higher blood lead levels and increased blood pressure (Schwartz et al., 2000c; Lee et al., 2001). Greater lead-associated reductions in renal function have been observed in adults with a variant allele of nitric acid synthetase, although cardiovascular outcomes, such as blood pressure and hypertension do not appear to depend on eNOS (endogenous nitric oxide synthase) allele (Weaver et al., 2003). Adults with variants of the hemochromatosis gene (C282Y and/or H63D) have higher patella lead levels (Wright et al., 2004). Only one polymorphism has been shown to modify lead neurotoxicity. Lead workers with the apolipoprotein E4 allele showed greater lead-associated decreases in neurobehavioral function than did workers with the E1, E2, or E3 alleles (Stewart et al., 2002). This work is in its early stages, and while it promises to shed light on bases of susceptibility to lead toxicity, firm conclusions cannot yet be drawn.

6.10.4.2 Influence of Nutritional Status on Risk

Only limited epidemiologic data are available on the role of nutritional status in modifying an individual's risk of lead toxicity. Adjusting for severity of environmental lead contamination, iron-deficient children appear to have higher blood lead levels than iron-replete children (Bradman et al., 2001). One interpretation of these data is that children experiencing the same external lead dose can experience different internal doses. In another study of iron status, a decline in blood lead level was associated with improved cognitive performance in iron-sufficient but not in iron-deficient children (Ruff et al., 1996). Among the possible explanations for this finding is that iron deficiency contributes to pharmacodynamic variability, increasing the

toxicity of a given lead dose. Some evidence suggests that the intellectual deficit associated with an elevated blood lead level is greater among undernourished children than well-nourished children (Gardner et al., 1998).

Several studies have suggested that dietary calcium may have a protective role by decreasing absorption of lead in the gastrointestinal tract and decreasing the mobilization of lead from bone stores to blood, especially during periods of high metabolic activity of the bone such as pregnancy and lactation. Lower calcium intake during pregnancy, especially the second half, appears to increase the mobilization of lead from bone compartments (Hernandez-Avila et al., 1996). However, in other studies, calcium supplementation had no effect on bone lead levels pregnant and lactating women (Rothenberg et al., 2000; Téllez-Rojo et al., 2002).

6.10.4.3 Influence of Health Status on Risk

The influence of an individual's health status on susceptibility to lead toxicity has been demonstrated most clearly for renal outcomes. Individuals with diabetes, hypertension, and chronic renal insufficiency are at increased risk of lead-associated declines in renal function and adverse effects have been demonstrated at blood lead levels below 5 µg/dL (Lin et al., 2001, 2003; Muntner et al., 2003; Tsaih et al., 2004). As discussed in an earlier section, children with nutritional deficiencies also appear to be more vulnerable to lead-associated neurobehavioral deficits.

6.10.4.4 Influence of Co-Exposures on Risk

Epidemiologic studies do not provide an adequate basis for determining whether cigarette smoking and/or alcohol affect the nature or severity of the health effects associated with lead exposure. Both factors have often been included in models of both child and adult health outcomes in order to adjust for potential confounding. In addition, both have been evaluated as pertinent pathways of adult exposure. However, their possible roles as effect modifiers have not been well studied.

Although most individuals are not exposed to lead in isolation but to lead in combination with other toxicants including cadmium, arsenic, mercury, and polychlorinated biphenyls, epidemiologic studies generally have focused solely on lead. Other toxicant exposures have sometimes been measured but are usually treated as potential confounders in the statistical

analyses, with their status as potential modifiers of lead toxicity left unexplored (Bellinger, 2000). As a result, epidemiologic studies do not provide an adequate basis for determining whether co-exposure to other toxicants affects the nature or severity of the health effects associated with lead exposure.

6.10.4.5 Influence of Timing of Exposure on Risk

6.10.4.5.1 Children

Studies do not provide a definitive answer to the question of whether lead-associated neurodevelopmental deficits are the result of exposure during a circumscribed critical period or of cumulative exposure. Although support can be cited for the conclusion that it is exposure within the first few postnatal years that is most important in determining long-term outcomes (Bellinger et al., 1992), other studies suggest that concurrent blood lead level is as predictive, and perhaps more predictive, of long-term outcomes than are early blood lead levels (Canfield et al., 2003a; Dietrich et al., 1993a,b; Tong et al., 1996; Wasserman et al., 2000b). Because of the complex kinetics of lead, an accumulative toxicant, it is extremely difficult to draw strong conclusions from these observational studies about windows of heightened vulnerability in children. The high degree of intraindividual "tracking" of blood lead levels over time, especially among children in environments providing substantial, chronic exposure opportunities (e.g., residence near a smelter or in older urban dwellings in poor repair), poses formidable obstacles to identifying the time interval during which exposure to lead caused the health effects measured in a study. It could be that damage occurred during a circumscribed period when the critical substrate was undergoing rapid development, but that the high correlation between serial blood lead levels impeded identification of the special significance of exposure at that time. Under such circumstances, an index of cumulative blood lead level or concurrent blood lead level, which might be a good marker of overall body burden under conditions of relatively steady-state exposure, might bear the strongest association with the adverse effect.

6.10.4.5.2 Aging Population

Increases in blood lead for postmenopausal women have been attributed to release of lead from the skeleton associated with increased bone remodeling during menopause in both occupationally- and environmentally-exposed women (Garrido-Latorre et al., 2003; Popovic et al., 2005). In middle-aged to elderly males from the Normative Aging Study, patella lead

accounted for the dominant portion of variance in blood lead (Hu et al., 1996). These findings provide evidence that the skeleton may serve as a potential endogenous source of lead in the aging population.

Considerable evidence also suggests that indicators of cumulative or long-term lead exposure are associated with adverse effects in several organ systems, including the central nervous, renal, and cardiovascular systems. Among occupationally-exposed men, higher tibia lead levels have been associated with increased cognitive decline over repeated assessments (Schwartz et al., 2005). With regard to the renal system, increased lead exposure may accelerate the effects of normal aging, producing a steeper age-related decline in function. Weaver et al. (2003) observed that higher lead exposure and dose were associated with worse renal function in older workers, but with lower blood urea nitrogen and serum creatinine in young workers.

6.10.4.5.3 Pregnancy

Potential mobilization of lead from the skeleton also occurs during pregnancy and lactation due to increased bone remodeling (Hertz-Picciotto et al., 2000; Manton, 1985; Silbergeld, 1991). In women who have been exposed to lead in childhood and have accumulated large stores in their bones, there may be significant mobilization of lead from bone to blood during late pregnancy and lactation. The greatest probability of lead toxicity for the mothers will be in postpartum while they are lactating; the infants will be particularly vulnerable during the prenatal period, especially in the last weeks of pregnancy (Manton et al., 2003).

A variety of adverse reproductive outcomes have been associated with higher paternal or maternal lead exposures, including reduced fertility, spontaneous abortion, gestational hypertension, congenital malformations, fetal growth deficits, and neurobehavioral deficits in offspring. The levels of exposure at which different adverse outcomes occur vary. Increased risks of spontaneous abortion, neurobehavioral deficits in offspring and, in some studies, gestational hypertension, have been reported at pregnancy blood lead levels below 10 μg/dL (Bellinger, 2005).

6.10.5 Reversibility of Lead Health Effects

6.10.5.1 Natural History of Effects

The absence of a clear operational definition of "reversibility" is a major impediment to drawing inferences about the natural history of any adverse effect associated with an accumulative neurotoxicant such as lead. Rather than indicating irreversibility, a performance deficit that remains detectable after external exposure has ended could reflect ongoing toxicity due to lead remaining at the critical target organ or lead deposited at the organ post-exposure as the result of redistribution of lead among body pools. As noted earlier, brain lead levels can remain elevated long after blood lead levels fall. A rigorous test of reversibility would require that every lead atom has been cleared from the body. This being unattainable, investigators must exploit opportunities that permit only weaker tests of hypotheses about reversibility. These include assessing the persistence of deficits previously associated with lead biomarkers and evaluating performance changes associated with natural experiments, i.e., events such as chelation or a change in external exposure that would be expected to perturb the equilibrium of lead among different body pools.

The likelihood of reversibility, as defined above, appears to be related, at least for the adverse effects observed in certain organ systems, to both the age-at-exposure and the age-at-assessment. In occupationally-exposed adults, the central and peripheral nervous system correlates of higher lead burdens appear to attenuate if exposure is reduced.

The prospective studies of childhood lead exposure, involving serial measurements of lead biomarkers and health outcomes, provide the best opportunities available to assess the natural history of adversities associated with low-level lead exposures. In some prospective studies, associations observed in infancy between biomarkers of prenatal exposure and neurodevelopment attenuated by the time children reached preschool age. It can be difficult to determine, however, whether this reflects actual disappearance of the effect or an increased difficulty in detecting it due to the emergence of associations between neurodevelopment and lead biomarkers measured postnatally. It is notable, however, that in some prospective studies of children, associations between biomarkers of prenatal lead exposure and various outcomes in middle adolescence have been reported, suggesting that the persistence of the associations might be endpoint-specific**.** For example, among children in Kosovo, Yugoslavia, IQ scores at the age of 8 years were inversely associated with a composite index of prenatal lead exposure (average

of mothers' blood lead levels at midpregnancy and at delivery) (Wasserman et al., 2000). This association was independent of changes in postnatal blood lead levels. Among 15 to 17 year old inner-city children in Cincinnati, OH, maternal blood lead levels in the 1st trimester (ranging from 1 to ~30 μg/dL) were inversely related to attention and visuoconstruction (Ris et al., 2004) and positively related to the frequency of self-reported delinquent behaviors (Dietrich et al., 2001).

The results of the prospective studies are more consistent in showing that higher postnatal lead biomarkers are associated with neurocognitive deficits that persist, in some studies, into early adulthood when the concurrent lead exposures are generally much lower. Ongoing external exposure does not appear to be necessary to maintain the deficits, although, as noted previously, it is not possible to exclude entirely a role for ongoing endogenous exposures of the target organs resulting from the redistribution, over time, of lead stores among different compartments. These data are consistent with those from experimental nonhuman primate studies, in which the temporal characteristics of exposure are manipulated as opposed to merely observed as in the human studies.

In most epidemiologic studies, the potential for true longitudinal analysis of the data has not been fully exploited, with the data evaluated in what is effectively a series of cross-sectional analyses.

6.10.5.2 Medical Interventions

Data from the Treatment of Lead Poisoned Children (TLC) study, a randomized controlled trial of the late outcomes of children treated for lead poisoning, support the hypothesis that the deficits associated with exposures of such magnitude are persistent and, possibly, permanent (Dietrich et al., 2004; Rogan et al., 2001). At 36-months post-treatment and at age 7 years, no significant differences in cognition or behavior were noted between the succimer and placebo groups. Current blood lead levels were significantly associated with cognitive performance at baseline, 36-months post-treatment, and at 7 years of age, and the regression coefficients were similar in magnitude to those estimated in observational studies (i.e., ~3 point IQ decline per 10 μg/dL increase in blood lead), providing a linkage between the results of the observational studies and those of this experimental study. However, within-child analyses

indicated that changes in developmental test scores over time were not consistently associated with changes over time in blood lead level.

6.10.6 Confounding of Lead Health Effects

6.10.6.1 Adjustment for Confounding in Epidemiologic Studies of Lead

The possibility that the adverse health effects associated with increased lead exposure in epidemiologic studies are, in fact, due to risk factors with which increased lead exposure is associated remains the most important impediment to drawing causal inferences. Various approaches have been taken to reduce the uncertainty this creates. Some investigators have specified the sampling frame or the eligibility criteria so as to increase the homogeneity of the study participants on factors known to be strong risk factors for the outcome of interest, thereby reducing the correlation between them and lead, and their potential to confound any association observed between increased lead exposure and poor outcome. Reducing confounding by means of design decisions has the disadvantage that an investigator cannot determine whether the impact of lead on the outcome varies depending on the factor whose range of potential values has been restricted. More frequently, however, investigators have relied on statistical procedures, applied post data collection, to identify and control for potential confounding. Unlike sample restriction, this approach preserves the opportunity to explore possible modification of the lead effect by cofactors.

Adjustment for confounding has been performed primarily using multiple regression analyses and data stratification. For multiple regression modeling, stepwise regression has been frequently used for covariate selection. Stepwise regression has many faults and is often less acceptable then the use of a few well-chosen covariates. However, the stepwise regression methodology may be considered less bias as it selects from a class of variables that represent a wide scientific viewpoint rather than the narrower one of the investigator. One problem with stepwise regression pointed out by Bellinger (2004) is that the usual adjustment strategy assumes that all the variance in the response shared by the exposure and the confounder belongs to the confounder. In some settings, this is likely to be excessively conservative, because confounders can, to some extent, also be proxies for exposure. This is further discussed in the next section.

Splitting the data set into smaller data sets (partitioning or stratification) and analyzing those data sets separately was used in some of the studies examining the relationship between

blood pressure and lead. This practice also has some advantages and disadvantages. An advanced statistical method could be used to determine how the partitioning should be done (Young and Hawkins, 1998), which could reveal relationships that would not be possible to detect using the usual regression techniques. A disadvantage of partitioning a small data set is that the smaller sample size may lack the power to detect otherwise detectable associations and to yield reliable estimates.

The segmented line model consists of joined straight line segments where the joined points are chosen to best fit the data (Quandt, 1958). The log-linear and the quadratic models have shown in several cases to better fit the biomarker-response relationship than the linear model. However, these models are not considered practicable for extrapolation outside the range of the biomarker variable. The segmented line model is suggested as a more reasonable model for extrapolation into the low-concentration sparse-data region.

6.10.6.2 Confounding Adjustment on Lead Health Effect Estimates

The ability of the investigator to determine how much of the apparent association between a lead biomarker and an outcome reflects residual confounding by a cofactor depends on the characteristics of the joint distribution of lead and the cofactor. Co-factors for lead health effects include maternal IQ, maternal smoking, alcohol use, birth weight, and many others depending on the health outcome of interest. Some of these cofactors are truly independent predictors and can be adjusted for using multiple regression analyses. Under some circumstances, however, lead and the cofactor may be so highly related that one cannot be confident that their associations with the outcome have been disentangled by the statistical methods applied. Moreover, the true causal relationships among lead, the cofactors, and the outcome might not be sufficiently well understood that the outcome variance shared by lead and the cofactors can be characterized appropriately in the analyses.

In studies of lead and neurodevelopment, the magnitude of the lead coefficient, reflecting the decline in test score per unit increase in the lead biomarker, is substantially reduced, often by half or more, by adjusting for markers of the social environment. During the 1980s, adjustment for parental IQ and quality of the home environment (e.g., HOME scores) became almost mandatory if the findings of a study of lead and children's cognitive outcomes were to be considered credible. While both factors surely strongly influence child outcomes in ways that

are independent of lead, a case can also be made that lead might contribute to the associations. A parent's IQ presumably reflects the parent's early lead exposure and, assuming that the physical environments in which a parent and child grow up are not completely unrelated to one another, provide similar lead exposure opportunities. Adjusting for parent IQ in evaluating the association between a child's lead exposure and his or her IQ, therefore, will result in an underestimate of the contribution of the child's lead exposure to his or her IQ. Similarly, if early lead exposure alters child behavior, the transactional model of child development would generate the prediction that the changes will elicit different behaviors from parents, altering the characteristics of the child rearing environment. For instance, increased lead exposure might result in an infant being more irritable, less soothable, and the parent less nurturing. In so far as measurement of the quality of the rearing environment in studies occurs after the children have experienced some lead exposure, the hypothesis that lead is responsible for shaping some aspects of that environment cannot be entirely dismissed, and control for HOME scores might be excessively conservative. For example, in the pooled analysis by Lanphear et al. (2005) that included seven prospective studies, the crude coefficient for concurrent lead and childhood IQ score was −4.66 (95% CI: −5.72, −3.60), but the coefficient adjusted for study site, HOME score, birth weight, maternal IQ, and maternal education was −2.70 (95% CI: −3.74, −1.66).

Other aspects of model building in assessing the association of lead with health outcomes also warrant comment. In many studies of lead and cognitive outcomes in children, investigators have adjusted for factors such as birth weight or length of gestation that might, themselves, reflect adverse effects of lead, i.e., mediating factors that lie between lead and condition on the causal pathway. The coefficient estimated for lead in a model that contained such factors would be smaller in magnitude than it would be if terms for such mediating factors had not been included.

Recognizing imperfections in the ability to measure such factors well, a concern is expressed that the lead coefficient could be reduced further, perhaps all the way to the null, if better, more comprehensive methods of measurement were applied. On the other hand, the methods used to adjust for such factors may be excessively conservative insofar as they attribute to a factor all of the outcome variance that it shares with lead, despite the likelihood that the true relationships among lead, social factors, and outcome are unlikely to be as simple as this model assumes. Some factors might, in part, be markers of lead exposure opportunities. For example,

both lead biomarker levels and lower cognitive function in children are associated with lower social class standing. Social class is a complex construct that conveys information about a multitude of factors that might influence children's health, including the amount of lead in environmental media. Thus, some of the association between lower social class and poorer health might reflect the effect of higher lead exposure. If so, routine adjustment of health outcome for social class in assessing the association between increased lead exposure and poorer health in children will fail to distinguish these lead-related and non-lead-related components of the association between social class and health, and, in fact, will assume that all of it is non-lead-associated. It is nearly impossible to actually determine if the problem of overadjustment exists in a particular data set. There are several statistical methods which attempt to address this problem. These include using partial F tests, ridge regression, path analysis, and structural equations. None of these methods are completely satisfactory.

6.10.7 Inferences of Causality

Even with more sophisticated and nuanced models, however, any conclusions about the causal forces generating the results of any observational epidemiologic study are necessarily uncertain. In the absence of random assignment to exposure group, residual confounding will always be a possible explanation of an observed association. As in other areas of epidemiology, a weight-of-evidence approach remains the best option available as a basis for drawing of causal inferences. If the association between a lead biomarker and a health outcome of interest is observed in settings that vary widely in terms of the characteristics of the social environment including sociodemographic and cultural characteristics, characteristics of the study participants, including nutritional status, genetic factors, and lifestyle factors, the likelihood that the association is attributable, in its entirety, to residual confounding is reduced. For instance, the pooled analyses of data contributed by many of the international prospective studies provide a compelling demonstration that the association between blood lead level and child IQ is remarkably robust across disparate socio-cultural settings (Lanphear et al., 2005). Even such consistency in the effect estimate across diverse settings is only indirect and weak evidence of causality, however. In general, epidemiologic studies rarely provide data that enhance our understanding of the “black box” between biomarkers of lead burden and indicators of health status. Epidemiologic data identify associations between exposure biomarkers and health

indicators, but are not highly informative regarding possible mechanisms of lead toxicity that underlie the associations. A critical stage in applying the overall weight-of-evidence approach is the examination of the epidemiologic data in the context of data from experimental animal behavioral and mechanistic studies. Although such data have their own limitations, they are not subject to many of the most important potential biases that can becloud the interpretation of the epidemiologic data.

6.10.8 Effects on the Individual Versus Effects on the Population

The critical distinction between population and individual risk, an issue pertinent to many questions in chronic disease epidemiology, has frequently been blurred in discussions of the public health implications of lead-associated decrements in health. With respect to neurodevelopment, while it may be true that a two- or three-point decline in IQ may not be consequential for an individual, the same level of decline observed in a population mean is of great importance. Similarly, although an increase of a few mm Hg in blood pressure may generally not be of concern for an individual's well-being, a very modest increase in the population mean is associated with substantial increases in the percentages of individuals with values that are sufficiently extreme that they exceed the criteria used to diagnose illness (Rose and Day, 1990). In other words, the mean value conveys substantial information about the percentage of individuals with clinically relevant, extreme values of the indicator. Moreover, interventions that shift the population mean by an amount that is without clinical consequence for an individual have been shown to produce substantial changes in the percentage of individuals with indicator values that are clinically significant (Bellinger, 2004). The following subsections will discuss quantitatively lead-related effects of a population level change in IQ and blood pressure.

6.10.8.1 Effects of Lead on Intelligence

The outcome most often examined to investigate neurotoxic effects of lead is IQ. Although the definition of "intelligence" is quite abstract, IQ remains a useful outcome measure as it is correlated with important measures of life success, such as academic achievement, earnings, and social status (Bellinger, 2003; Weiss, 2000). Several studies reported quantitative relationships between full scale IQ and current blood lead levels for children aged 5 to 11 years

old, and these are summarized in Table 6-10.1. The estimated relationships as reported by the authors are used.

Table 6-10.1. Summary of Studies with Quantitative Relationships for IQ and Blood Lead

Reference	Study Location	n	Estimated Slope (IQ points/µg/dL) – Blood Lead 10th to 90th Percentile	Estimated Slope (IQ points/µg/dL) – Blood Lead Under 10 µg/dL
Bellinger et al. (1992)	Boston, Massachusetts	116	−0.5	NA
Canfield et al. (2003a)	Rochester, New York	182	−0.7	−0.8
Dietrich et al. (1993a)	Cincinnati, Ohio	221	−0.3	−0.3
Ernhart et al. (1989)	Cleveland, Ohio	160	−0.1	NA
Wasserman et al. (1997)	Kosovo, Yugoslavia	231	−0.2	NA
Baghurst et al. (1992)	Port Pirie, South Australia	324	−0.2	−0.4
Silva et al. (1988)	Dunedin, New Zealand	579	−0.3	−0.3
Lanphear et al. (2005)	International Pooled Analysis	1,333	−0.5	−0.2

The curves over a range of blood lead levels from the 10th percentile to the 90th percentile are shown in Figure 6-10.2. The curves are restricted to that range because log-linear curves become very steep at the lower end of the blood lead levels, and this may be an artifact of the model chosen. The percentiles are estimated using various methods and are only approximate values. Studies which estimated a linear relationship are shown as reported, and similarly for the log-linear relationships. Note that these are not forest plots of slopes or hazard ratios – they are the actual estimated relationships.

The analysis by Lanphear et al. included the studies of Baghurst et al. (1992), Bellinger et al. (1992), Canfield et al. (2003a), Dietrich et al. (1993a), Ernhart et al. (1989) and Wasserman et al. (1997). The pooled analysis also included the Mexico City study of Schnaas et al. (2000). The results from Schnaas et al. are not included in Table 6-10.1 or Figure 6-10.2 because the

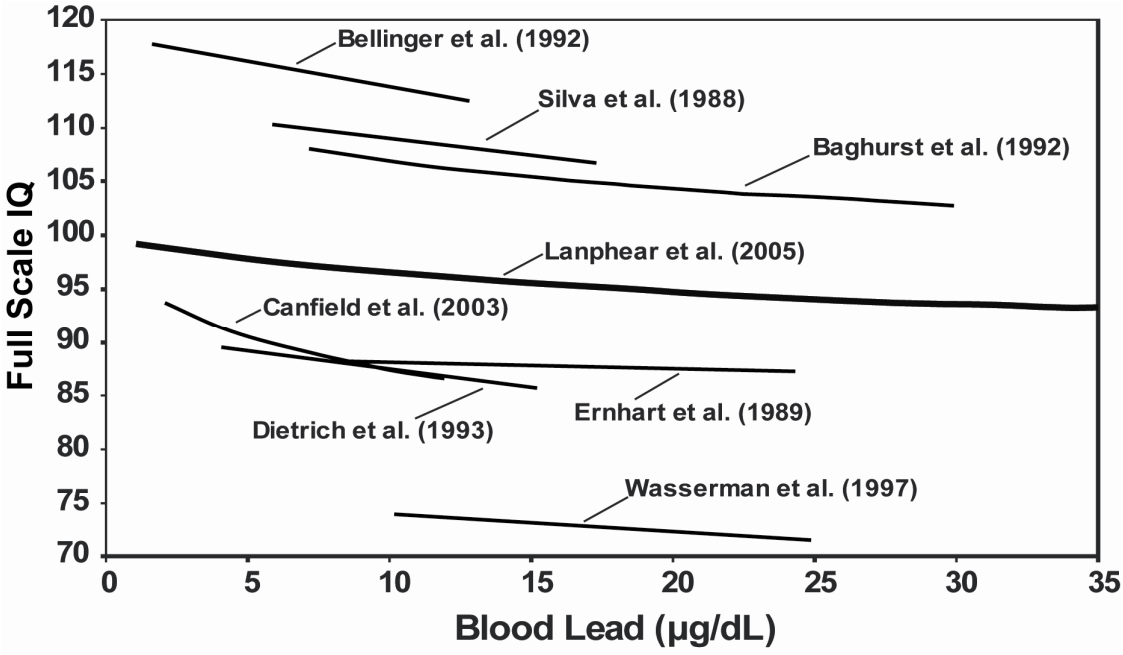

Figure 6-10.2. Concentration-response relationships of IQ to blood lead for the individual studies and the pooled analysis by Lanphear et al. (2005).

authors did not provide regression coefficients in their paper, thus concentration-response relationship were not estimable. The study by Silva et al. (1988) is not included in the pooled analysis of Lanphear et al., but is included in this section as its results are comparable and informative.

Several conclusions can be drawn from these graphs. First, note that the overall IQ levels are quite different. This results from different populations and from different applications of the IQ tests. Second, all studies showed a decreasing IQ score as the blood lead level increased. It is the slope of the studies that is relevant, not the actual IQ scores. Third, for studies with lower blood lead levels, the slopes appear to be steeper. This is the reason that many authors choose to use the log-linear model. However, for those studies where the blood leads were generally high, the log-linear and linear models are almost identical. Thus it is not surprising that some authors chose a linear model instead of a log-linear model. The curves in Figure 6-10.2 do not show evidence of a no-effect threshold because the slopes increase as the blood lead levels become smaller. The observed mean adjusted IQ levels (for blood lead <5, 5 to 10,

10 to 15, 15 to 20, and >20 µg/dL) reported by Lanphear et al. (2005) also show no evidence of a threshold, as seen in Figure 6-10.3.

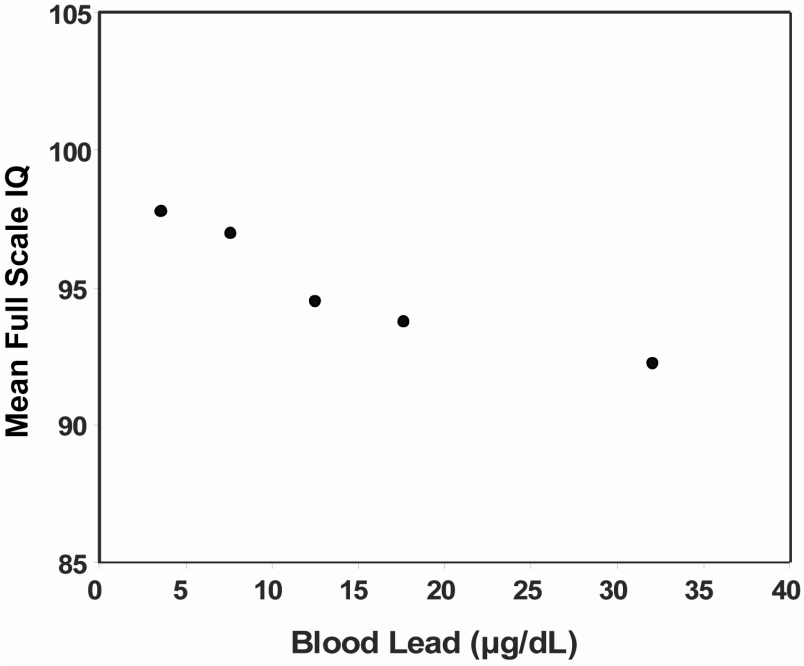

Figure 6-10.3. Mean blood lead levels adjusted for HOME Score, maternal education, maternal IQ, and birth weight from the pooled analysis of seven studies by Lanphear et al. (2005). Mean adjusted IQ levels at blood lead levels of <5, 5 to 10, 10 to 15, 15 to 20, and >20 µg/dL are shown.

Weiss (1990) predicted, on purely statistical grounds, that a downward shift of five points in mean IQ, if the amount of dispersion in the distribution remained the same, should be accompanied by a doubling of the numbers of individuals with scores two or more standard deviations below the mean and a reduction by half of the number of individuals with scores two or more standard deviations above the mean. With respect to lead, the general accuracy of this prediction has been empirically demonstrated in two different datasets by Needleman et al. (1982) and Bellinger (2004). The example below provides further evidence of the change in percentages of individuals with IQ <70 or <50 points after restricting the analysis to those with blood lead levels less than 10 µg/dL.

The average slope was estimated for those studies with a significant portion of the subjects with blood lead levels less than 10 µg/dL. These average slopes are given in Table 6-10.1. In addition, the results of Lanphear et al. (2005) were considered. The average slope at blood lead levels less than 10 µg/dL from that pooled analysis was −0.5 IQ points per µg/dL. Based on the individual studies and the pooled analysis it appears that the average slope is between −0.3 and −0.5 points per µg/dL, with the exception of the large negative slope of −0.8 points per 10 µg/dL from the study by Canfield et al. (2003a). The value of −0.4 points per µg/dL will be used in calculations of the implications of the slope at blood lead levels less than 10 µg/dL.

A nonexposed population was assumed to have a standard mean IQ of 100 and standard deviation of 15 at a blood lead exposure of 0 µg/dL. The fraction of the population that would have an IQ <70 or <50 as a function of blood lead level was then calculated. The results are shown in Figure 6-10.4. Note that the fraction with an IQ level below 70, a level often requiring community support to live (World Health Organization, 1992) increases from a little over 2 percent for no lead exposure to about 4 percent with a blood lead level of 10 µg/dL. In addition, the fraction with an IQ level below 50, a level often requiring continuous support to live (World Health Organization, 1992) increases from a little over 4 per 100,000 for no lead exposure to about 11 per 100,000 with a blood lead level of 10 µg/dL.

A shift in the mean value of a health indicator has substantial importance for both extremes of the distribution. In the case of lead, a downward shift in the mean IQ value is associated not only with a substantial increase in the percentage of individuals achieving very low scores, it is associated as well with a substantial decrease in the percentage achieving very high scores. Based on the study by Bellinger et al. (1987) examining intelligence test scores of lead-exposed children, Weiss (1988) discussed the shift of the population distribution of IQ from a mean of 100 and a standard deviation of 15 to a mean of 95, a 5% reduction. When the mean IQ level is 100, 2.3% of the individuals in a given population would score above 130. However, with the population distribution shift and the resulting mean decline in IQ, only 0.99% of the individuals would score above 130. Weiss states that the implication of such as loss transcends the current circumscribed definitions of risk.

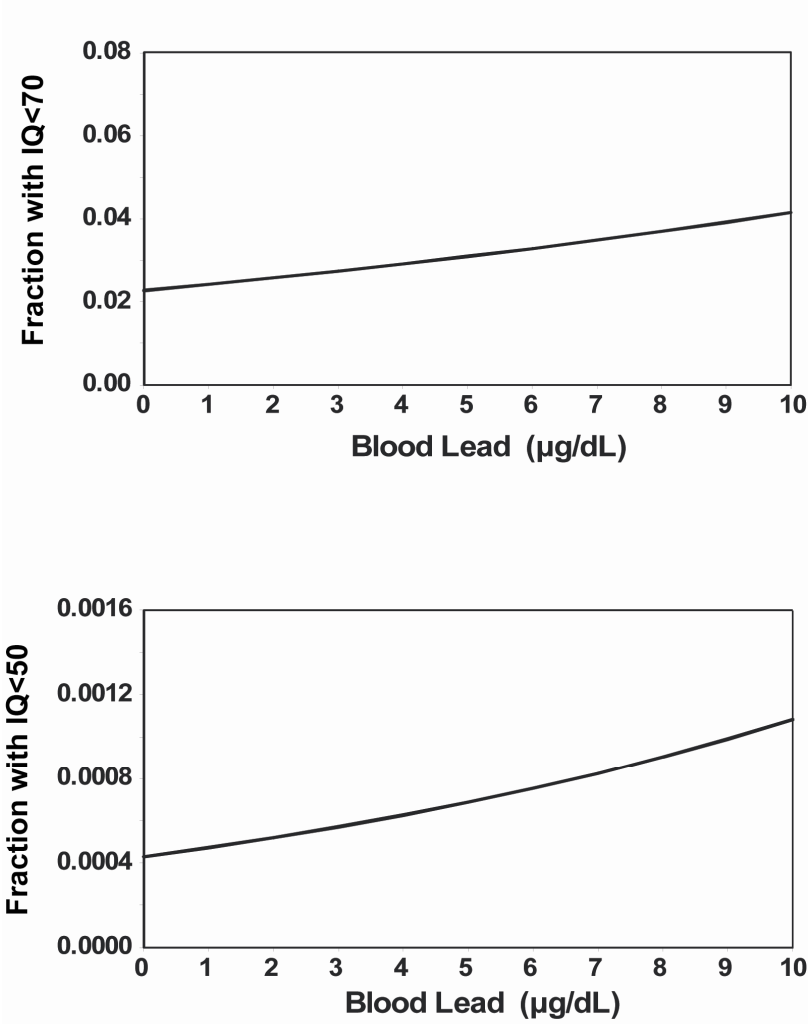

Figure 6-10.4. Effect of blood lead on fraction of population with IQ level <70 or <50 points.

6.10.8.2 Cardiovascular Effects of Lead

In studies investigating the cardiovascular effects of lead, blood pressure has been examined most frequently. Results from the Framingham Heart Study show that higher levels of blood pressure, even within the nonhypertensive range, impose increased rates of cardiovascular disease (Kannel, 2000a,b). A continuous graded increase in cardiovascular risk is observed as blood pressure increases, with no evidence of a threshold value. Most events arise not in the most severe cases, but mainly in those with high normal blood pressure (i.e., mild hypertension). This view is further supported by the Seventh Report of the Joint National Committee on

Prevention, Detection, Evaluation, and Treatment of High Blood Pressure (Chobanian et al., 2003). Kannel (2000b) states that reducing even moderate elevation in blood pressure is likely to be beneficial.

Kannel (2000a) states that systolic blood pressure exerts a strong, influence on cardiovascular events, as it is the prime causal function of hypertension and its adverse cardiovascular sequelae. Cardiovascular events include coronary disease, stroke, peripheral artery disease, and cardiac failure. Risk ratios are larger for cardiac failure and stroke, but coronary disease (i.e., myocardial infarction, angina pectonis, sudden death) is the most common and most lethal sequela of hypertension (Kannel, 1996). Kannel (2000a) notes that the Framingham Heart Study has recognized that elevated blood pressure tends to occur alongside other major risk factors of cardiovascular disease such as glucose intolerance, dyslipidemia, abdominal obesity, and left ventricular hypertrophy, among others. If a cluster of multiple risk factors is present, the hazard is formidable for coronary disease and stroke.

No critical level of blood pressure is evident. The risk appears to be simply proportional from the lowest to the highest level recorded. In the Multiple Risk Factor Intervention Trial (MRFIT), Neaton et al. (1995) confirmed a continuing and graded influence of systolic blood pressure on cardiovascular disease mortality extending down into the range of <140 mm Hg. The Prospective Studies Collaboration (2002) meta-analysis of 61 prospective studies relates blood pressure to vascular mortality without indication of a threshold down to 115/75 mm Hg. The absence of a demonstrable safe or critical level of blood pressure suggests using the range of blood pressure rather than discrete categories such as hypertension.

Many studies have suggested a relationship between blood lead and systolic blood pressure. In particular, the meta-analysis of Nawrot et al. (2002) indicated that a doubling of the blood lead corresponded to a 1 mm Hg increase in systolic blood pressure. Although this magnitude of increase is not clinically meaningful for an individual, a population shift of 1 mm Hg is important.

The Framingham Heart Study results (Kannel, 2000a) were used to estimate a typical population distribution of systolic blood pressure values (Figure 6-10.5). The distribution of systolic blood pressure values was approximated well by a lognormal distribution for both women and men ($p \geq 0.4$). The relationship between systolic blood pressure and the risk of

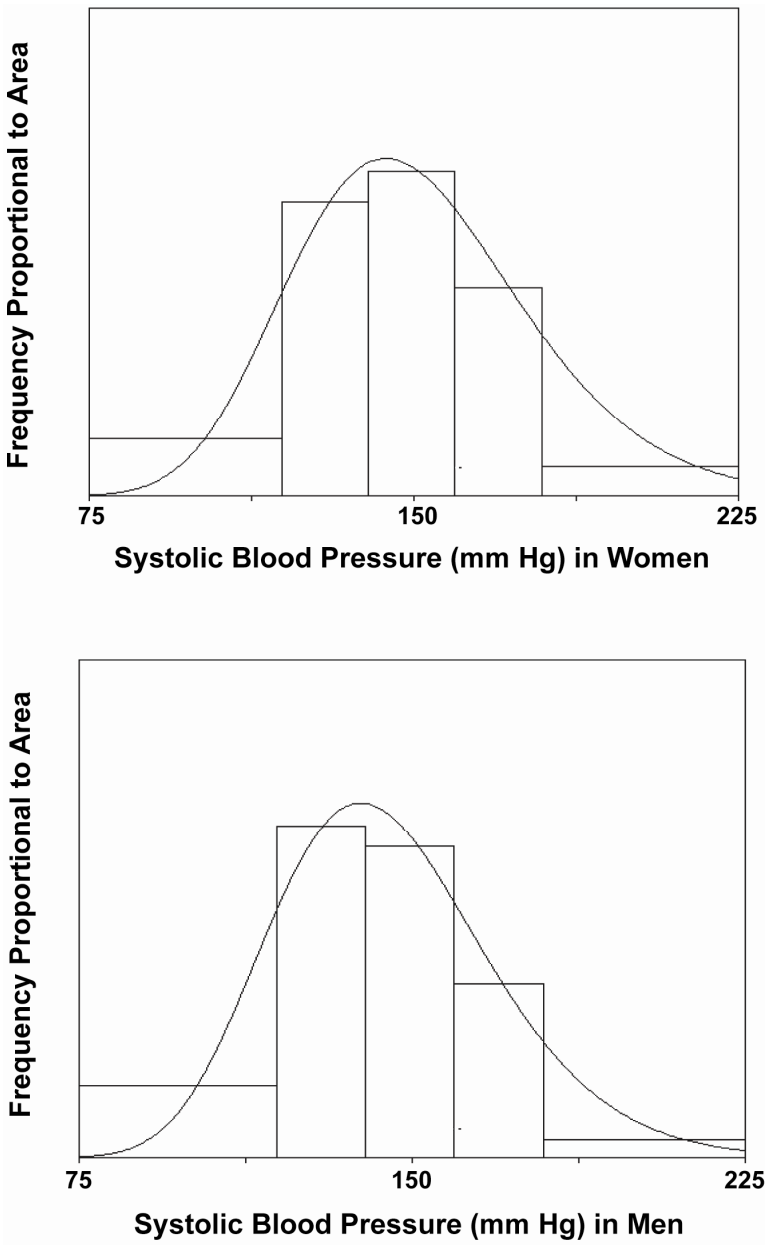

Figure 6-10.5. Distribution of systolic blood pressure in women and men aged 35 to 64 years from the Framingham Heart Study (Kannel, 2000a).

cardiovascular events was also given by Kannel (2000a). The relationships are shown in Figure 6-10.6.

To estimate population risk, it was assumed that the effect of blood lead on blood pressure was to shift the entire distribution by the amount given by Nawrot et al. (2002). For each shift in

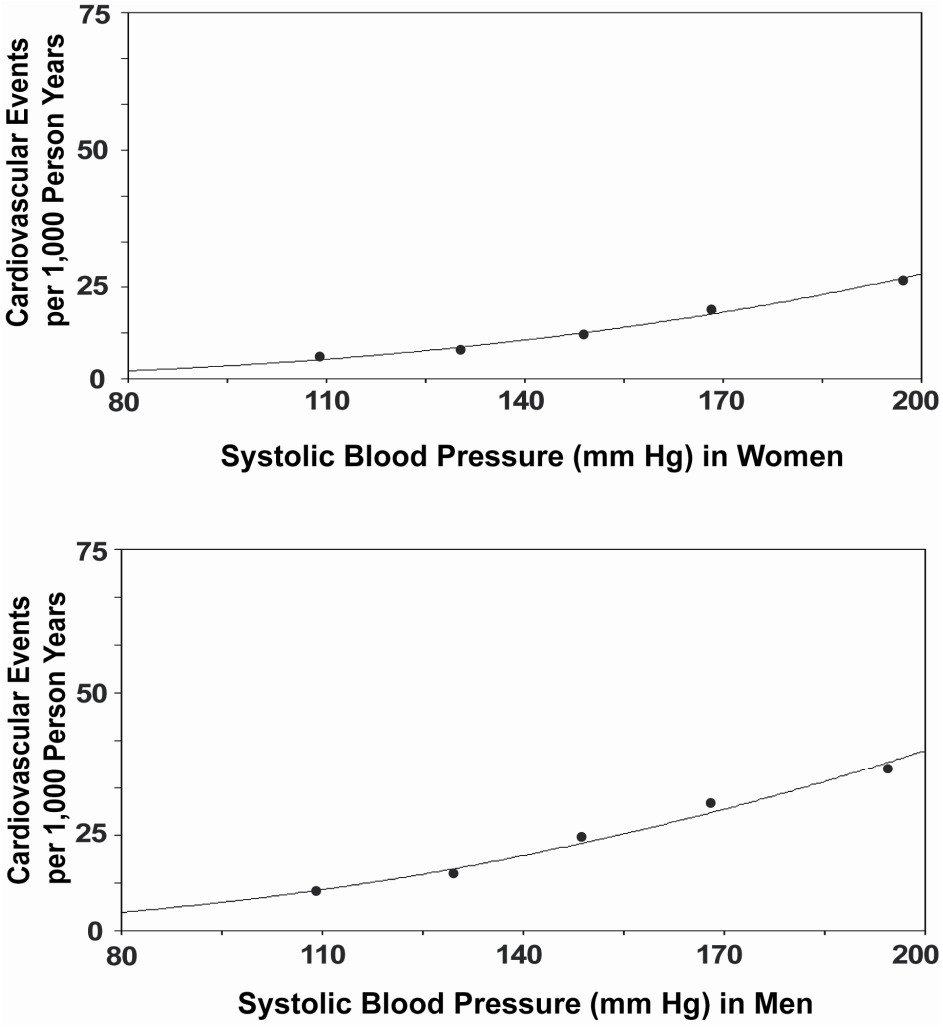

Figure 6-10.6. Relationship of cardiovascular events (coronary disease, stroke, peripheral artery disease, cardiac failure) to systolic blood pressure in women and men aged 35 to 64 years from the Framingham Heart Study (Kannel, 2000a).

the distribution, the entire distribution was integrated out over the risk given in Figure 6-10.6. The result estimated was expected number of cardiovascular events per 1,000 person years, and this was plotted for blood lead levels ranging from 5 to 15 µg/dL for both women and men. The results are shown in Figure 6-10.7. Although the effects are modest, they translate into a large number of events for a moderate population size. For example, a decrease in blood lead from 10 to 5 µg/dL results in an annual decrease of 27 events per 100,000 women and 39 events per 100,000 men.

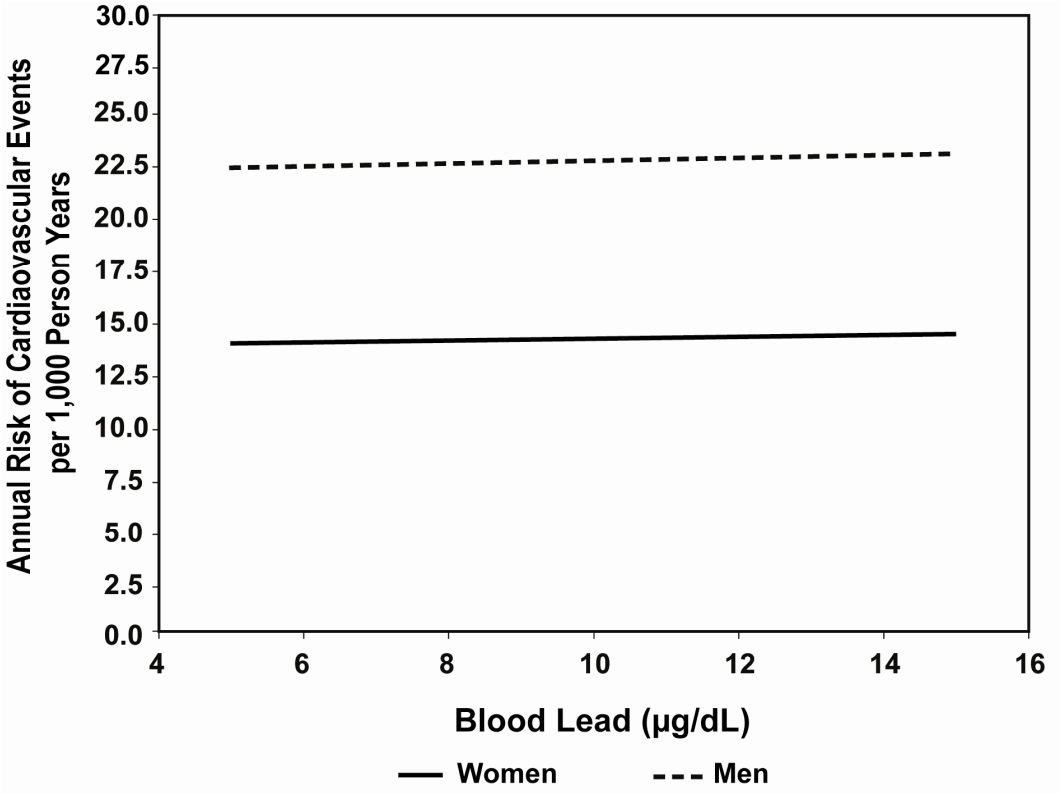

Figure 6-10.7. Effect of blood lead on expected annual risk of cardiovascular events per 1,000 person years.

6.10.9 Summary of Key Findings and Conclusions Derived from Lead Epidemiology Studies

The remarkable progress that has been made since the mid-1980s in understanding the effects of lead on health can be gauged by noting the changes that have occurred over time in the questions investigators have addressed. In the 1980s, the question of interest was often, "Does low-level lead exposure affect health?" The questions asked in recent studies have more often focused on details of the associations, including the shapes of concentration-response relationships, especially at levels well within the range of general population exposures, biological and socio-environmental factors that either increase or decrease an individual's risk, the prognoses associated with lead-associated effects, the efficacy of interventions to reduce adverse effects, and so on. In fact, "low-level," a term long-used to describe exposures that are not sufficiently high to produce clinical signs and symptoms, is increasingly being recognized as

a descriptor that has little biological meaning and is interpretable only in a specific historical context. What was considered "low" in the 1980s is an order of magnitude higher than the current mean level in the U.S. population, and the current mean remains perhaps as much as two orders of magnitude above "natural" background levels in humans. The current CDC screening guideline for children of 10 µg/dL is not a "bright line" separating toxicity from safety, but merely a risk management tool. There is no level of lead exposure that can be clearly identified, with confidence, as "safe." Recent studies of lead neurotoxicity in infants have observed adverse effects at blood lead levels of only 1 or 2 µg/dL and adverse renal outcomes have been reported at blood lead levels below 5 µg/dL. Public health interventions have resulted in declines, over the last 25 years, of more than 90% in the mean blood lead level within all age and gender subgroups of the U.S. population, substantially decreasing the numbers of individuals at risk of lead toxicities.

The following are a listing of key health outcomes discussed earlier in the epidemiology chapter:

- **Neurotoxic effects of lead in children.** The effects of lead on neurobehavior in children have been observed with remarkable consistency across numerous studies of various designs, populations, and developmental assessment protocols. The negative impact of lead on neurocognitive ability and other neurobehavioral outcomes persist in most recent studies even after adjustment for numerous confounding factors including social class, quality of caregiving, and parental intelligence. An international pooled analysis of seven prospective cohort studies offers evidence that exposure to lead has an effect on the intellectual attainment of preschool and school age children even at blood lead levels below 10 µg/dL.

  Epidemiologic studies have demonstrated that lead also may be associated with increased risk for antisocial and delinquent behavior, which may be a consequence of attention problems and academic underachievement among children who have suffered higher exposures to lead during their formative years. Direct measures of brain damage using Magnetic Resonance Imaging (MRI) and Magnetic Resonance Spectroscopy (MRS) also are suggesting evidence of harm due to lead exposure. Pharmacological or nutritional intervention strategies generally have not shown to eliminate or reduce lead-associated neurodevelopmental morbidities.

- **Neurotoxic effects of lead in adults.** Environmental lead exposure has not been found to be associated with impaired cognitive performance in the elderly if competing risk factors are considered. In adults, the effect of lead on the nervous system may not be detected through neurobehavioral testing due to cognitive reserve, the ability to compensate for brain impairment.

  Numerous studies of occupational lead exposure observed associations of blood lead with peripheral sensory nerve impairment, visuomotor and memory impairment, and postural sway abnormalities. Past occupational exposure to lead also was associated with increased risk of developing Amyotrophic Lateral Sclerosis (ALS), motor neuron disease, and essential tremor. The odds of developing ALS and essential tremor were significantly increased in individuals with the ALAD2 allele. These neurobehavioral impairments in occupationally-exposed individuals were typically associated with higher blood lead levels (approximately 30-40 μg/dL); however, essential tremor was found to be associated with much lower blood lead levels (mean 3 μg/dL).

- **Renal effects of lead.** In the general population, both cumulative and circulating lead was found to be associated with longitudinal decline in renal functions. In the large NHANES III study, renal dysfunction was observed in hypertensives at a mean blood lead of only 4.2 μg/dL. These results provide strong evidence that the kidney is a target organ for adverse effects from lead in adults at current U.S. environmental exposure levels. The renal impact in children environmental lead exposure is difficult to assess since the most studies have measured early biological effect markers and their prognostic value is uncertain.

  Studies involving the longitudinal assessment of renal function decline in susceptible patient populations observed that low levels of blood lead (<5 μg/dL) and chelatable lead levels were associated with decline in glomerular filtration rate over a 4 year follow-up period in patients with chronic renal insufficiency. Renal function in these patients was found to stabilize and, in some cases, improve after therapeutic chelation.

- **Cardiovascular effects of lead.** Epidemiologic studies support the relationship between increased lead exposure and increased adverse cardiovascular outcome, including increased blood pressure and increased incidence of hypertension. A recent meta-analysis reported that a doubling of blood lead level was associated with a 1.0 mm Hg increase in systolic blood pressure and a 0.6 mm Hg increase in diastolic pressure. Studies also have found that cumulative past lead exposure (e.g., bone lead) may be as important, if not more, than present exposure in assessing cardiovascular effects. The evidence for an association of lead with cardiovascular morbidity and mortality is limited but supportive.

- **Reproductive and developmental effects of lead.** The epidemiologic evidence suggests small associations between exposure to lead and male reproductive outcomes, including perturbed semen quality and increased time to pregnancy. These associations appear at blood lead levels greater the 45 μg/dL, as most studies only considered exposure in the occupational setting. There are no adequate data to evaluate associations between lead exposure and female fertility. For many other outcomes, the observed associations are fairly small, especially at the levels of exposure that are currently of interest. However, there may be populations that are highly susceptible to lead-related reproductive effects, especially if they have additional risk factors for these outcomes.

- **Genotoxic and carcinogenic effects of lead.** Studies of genotoxicity consistently find associations of lead exposure with DNA damage and micronuclei formation; however, the associations with the more established indicator of cancer risk, chromosomal aberrations, are inconsistent. Epidemiologic studies of highly-exposed occupational populations suggest a relationship between lead and cancers of the lung and the stomach; however the evidence is limited by the presence of various potential confounders, including coexposures (e.g., arsenic, cadmium), smoking, and dietary habits. The 2004 IARC review concluded that lead was a probable carcinogen based on limited evidence in humans and sufficient evidence in animals.

- **Effects of lead on the immune system.** Several studies have examined possible associations between lead exposures and biomarkers of immune function. Findings from recent epidemiologic studies suggest that lead exposure may be associated with effects on

cellular and humoral immunity. These effects include changes in serum immunoglobulin levels; perturbation of peripheral lymphocyte phenotype profiles, including decreases in peripheral blood T-cell abundance and changes in T-cell to B-cell abundance ratios; suppression of lymphocyte activation; and suppression of neutrophil chemotaxis and phagocytosis. Studies of biomarkers of humoral immunity in children have consistently found significant associations between increasing blood lead concentrations and serum IgE levels at blood lead levels below 10 µg/dL.

- **Effects of lead on the hematopoietic system.** Lead exposure has been associated with disruption of heme synthesis in both children and adults. Increases in blood lead concentration of approximately 20–30 µg/dL are sufficient to halve erythrocyte ALAD activity and sufficiently inhibit ferrochelatase to double erythrocyte protoporphyrin levels. Perturbation of erythropoiesis, indicated by changes in serum erythropoietin and progenitor cells, occurs in the absence of detectable changes in blood hemoglobin levels or hematocrit in children and adults at blood lead levels below 40 µg/dL. Risk of clinical anemia in children becomes appreciable at much higher blood lead concentrations.

- **Effects of lead on the hepatic and gastrointestinal system.** Studies of hepatic enzyme levels in serum suggest that liver injury may be present in lead workers; however, associations specifically with lead exposures are not evident. Studies of urinary metabolites of cytochrome P450 phenotypes CYP2A6 and CYP3A4 suggest possible associations between lead exposure and suppression of hepatic enzyme activity in adults and children. Several studies observed an association between occupational lead exposure and prevalence of symptoms of gastrointestinal colic. These hepatic and gastrointestinal effects are largely observed only at blood lead concentrations (>40 µg/dL).

- **Effects of lead on the endocrine system.** Most studies have yielded no associations, or weak associations, of lead exposure with thyroid hormone status and male reproductive endocrine status in highly-exposed occupational populations. Children exposed to relatively high levels of lead (blood lead >30 µg/dL) exhibit depressed levels of circulating 1,25-dihydroxy vitamin D (1,25-OH-D). However, associations between serum vitamin D

status and blood lead were not evident in a study of calcium-replete children who had average lifetime blood lead concentrations below 25 μg/dL.

- **Effects of lead on bone and teeth.** The epidemiologic evidence is limited, but suggestive of an association between lead exposure and bone toxicity. Studies have found an association between occupational exposure to lead and Paget's disease. However, it is difficult to assess whether increased lead results from bone diseases or the bone disease is a result of increase lead exposure. Increased risk of dental caries has been associated with lead exposure in children and adults. Lead effects on caries were observed in populations whose mean blood lead levels were less than 10 μg/dL.

- **Effects of lead on ocular health.** Recent longitudinal studies provide evidence for possible associations between lead exposure and adverse ocular health outcomes in low- to moderately-exposed populations. In children whose mothers had blood lead levels of 10.5–32.5 μg/dL in mid-pregnancy, an association was observed between lead exposure and visual evoked retinal responses. Middle-aged males whose tibia bone lead levels were 31–126 μg/g had increased risk of cataracts.

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8. ENVIRONMENTAL EFFECTS OF LEAD

8.1 TERRESTRIAL ECOSYSTEMS

8.1.1 Introduction

Surface soils across the United States are enriched in lead (Pb) relative to levels expected from natural (geogenic) inputs (Erel and Patterson, 1994; Francek, 1992; Friedland et al., 1984; Marsh and Siccama, 1997; Murray et al., 2004; Yanai et al., 2004). While some of this contaminant Pb is attributed to paint, salvage yards, and the use of Pb arsenate as a pesticide in localized areas (Francek, 1997), Pb contamination of surface soils is essentially ubiquitous because of atmospheric pollution associated with waste incineration, the combustion of fossil fuels, and metal smelting and production (Newhook et al., 2003; Polissar et al., 2001). However, lead inputs to terrestrial ecosystems in the United States have declined dramatically in the past 30 years. The primary reason for this decline has been the almost complete elimination of alkyl-lead additives in gasoline in North America. Also, emissions from smelters have declined as older plants have been shut down or fitted with improved emissions controls.

Lead released from forest floor soils in the past has been largely immobilized in mineral soils (Miller and Friedland, 1994; Johnson et al., 1995b, 2004; Johnson and Petras, 1998; Watmough et al., 2004). The amount of Pb that has leached into the mineral soil appears to be on the order of 20 to 50% of the total anthropogenic Pb deposition.

Most terrestrial ecosystems in North America remain sinks for lead, despite reductions in atmospheric Pb deposition of more than 95%. Although inputs of Pb to ecosystems are currently low, Pb export from watersheds via groundwater and streams is substantially lower. Reported concentrations of Pb in waters draining natural terrestrial ecosystems have always been low (Bacon and Bain, 1995; Johnson et al., 1995b; Wang et al., 1995; Vinogradoff et al., 2005), generally less than 1 ng L^{-1}, even at moderately polluted sites (Laskowski et al., 1995). Therefore, even at current input levels, watersheds are accumulating industrial Pb.

The current chapter summarizes the most relevant information from the 1986 Air Quality Criteria Document (AQCD) and reviews new information that has become available on the potential effects of atmospheric lead inputs on the terrestrial ecosystem. It has been organized to address: methodologies used in terrestrial ecosystem research (Section 8.1.2); the distribution of

atmospherically delivered lead in terrestrial ecosystems (Section 8.1.3); lead uptake and mechanisms of action (Section 8.1.4); toxic effects of lead on terrestrial organisms (Section 8.1.5); and, lead effects on natural terrestrial ecosystems (Section 8.1.6). The major conclusions and recommendations from each of these subject areas are summarized here.

8.1.1.1 Methodologies in Terrestrial Ecosystem Research

Several methodologies used in terrestrial ecosystems research are described in Sections 8.1.2 and 8.1.3. One of the key factors necessary for understanding ecological risks is related to bioavailability. The National Research Council (NRC) 2002 review on bioavailability defined the "bioavailability processes" in terms of three key processes. One of these processes, contaminant interactions between phases, is more commonly referred to as "speciation." For a given metal or metalloid, the term speciation describes the chemical's ability to interact with its biological or chemical surroundings by characterizing its physicochemical properties that are relevant to bioavailability.

Methods to address bioavailability (speciation), and methods used to reduce Pb bioavailability, are summarized in this section.

Analytical Tools and Models

A wide variety of analytical tools have been used to characterize a metal's speciation as it is found in various media:

- XRD - X-ray diffraction;
- EPMA - electron probe microanalysis;
- PIXE and μPIXE - particle induced X-ray emission;
- XPS - X-ray photoelectron spectroscopy;
- XAS - X-ray absorption spectroscopy;
- SIMS - secondary ion mass spectrometry;
- sequential extractions; and,
- single chemical extractions.

EMPA techniques provide the greatest information on metal speciation. Other techniques, such as EXAFS (extended X-ray absorption fine structure) and EXANES (extended X-ray

absorption near edge spectroscopy), show great promise and will be important in solving key mechanistic questions.

The tools that have been used most often to evaluate speciation for metal particles in solution include the following computer-based models: MINTEQL, REDEQL2, ECOSAT, MINTEQA2, HYDRAQL, PHREEQE, and WATEQ4F.

Metal Speciation for Plants

When considering the bioavailability of a metal to plants from soils and sediments, it is generally assumed that both the kinetic rate of supply and the speciation of the metal to either the root or shoot are highly important. In soils and sediments, generally only a small volume of water is in contact with the chemical form, and although the proportion of a metal's concentration in this pore water to the bulk soil/sediment concentration is small, it is this phase that is directly available to plants. Therefore, pore water chemistry (i.e., metal concentration as simple inorganic species, organic complexes, or colloid complexes) is most important.

Tools currently used for metal speciation for plants include (1) in situ measurements using selective electrodes (Gundersen et al., 1992; Archer et al., 1989; Wehrli et al., 1994); (2) in situ collection techniques using diffusive equilibrium thin films (DET) and diffusive gradient thin films (DGT) followed by laboratory analyses (Davison et al., 1991, 1994; Davison and Zhang, 1994; Zhang et al., 1995); and (3) equilibrium models (SOILCHEM) (Sposito and Coves, 1988).

Influence of Soil Amendments on Bioavailability

The removal of contaminated soil to mitigate exposure of terrestrial ecosystem components to Pb can often present both economic and logistic problems. Because of this, recent studies have focused on in situ methodologies to lower soil-Pb RBA (Brown et al., 2003a,b). To date, the most common methods studied include the addition of soil amendments in an effort to either lower the solubility of the Pb form or to provide sorption sites for fixation of pore-water Pb. These amendments typically fall within the categories of phosphate, biosolid, and Al/Fe/Mn-oxide amendments.

Phosphate amendments have been studied extensively and, in some cases, offer the most promising results (Brown et al., 1999; Ryan et al., 2001; Cotter-Howells and Caporn, 1996; Hettiarachchi et al., 2001, 2003; Yang et al., 2001; Ma et al., 1995). A number of potentially

significant problems associated with phosphate amendments have been recognized, including both phyto- and earthworm toxicity (Ownby et al., 2005; Cao et al., 2002; and Rusek and Marshall, 2000). Both of these toxicities are primarily associated with very high applications of phosphorous and/or decreased soil pH. Indications of phytotoxicity are often balanced by studies such as Zhu et al. (2004) that illustrate a 50 to 70% reduction in shoot-root uptake of Pb in phosphate-amended soils. Additionally, the added phosphate poses the potential risk of eutrophication of nearby waterways from soil runoff.

Biosolids have been used historically in the restoration of coal mines (Haering et al., 2000; Sopper, 1993). More recently, workers have demonstrated the feasibility of their use in the restoration of mine tailings (Brown et al., 2000), and urban soils (Brown et al., 2003a; Farfel et al., 2005). As with phosphate amendments, problems with biosolid application have also been documented. Studies have shown that metal transport is significantly accelerated in soils amended with biosolids (Al-Wabel et al., 2002; McBride et al., 1999, 1997; Lamy et al., 1993; Richards et al., 1998, 2000).

8.1.1.2 Distribution of Atmospherically Delivered Lead in Terrestrial Ecosystems

Advances in technology since the 1986 AQCD have allowed for a more quantitative determination of the mobility, distribution, uptake, and fluxes of atmospherically-delivered Pb in terrestrial ecosystems.

Lead Speciation in Solid Phases

Selective chemical extractions have been employed extensively for quantifying amounts of a particular metal phase (e.g., PbS, Pb-humate, Pb-Fe, Mn oxide) present in soil rather than total metal concentration. Selective extractions can be a relatively rapid, simple, and inexpensive means for determining metal phases in soils, and the generated data can be linked to potential mobility and bioavailability of the metal (Tessier and Campbell, 1987). However, some problems persist with the selective extraction technique. First, extractions are rarely specific to a single phase. For example, while peroxide (H_2O_2) is often used to remove metals bound in organic matter in soils, some researchers have demonstrated that this reagent destroys clay minerals and sulfides (Ryan et al., 2002). Peroxide solutions may also be inefficient at removing metals bound to humic acids, and in fact could potentially result in the precipitation of metal-

humate substances. In addition to non-selectivity of reagents, significant metal redistribution has been documented during sequential chemical extractions (Ho and Evans, 2000), and many reagents may not extract targeted phases completely. Therefore, while chemical extractions do provide some useful information on metal phases in soil, the results should be treated as "operationally defined," e.g., "H_2O_2 liberated-Pb" rather than "Organic Pb."

Synchrotron radiation (X-rays) allows researchers to probe the electron configuration of metals in untreated soil samples. Since different elements have different electron binding energies, X-rays can be focused in an energy window specific to a metal of interest. The precise energy required to dislodge a core electron from a metal will be a function of the oxidation state and covalency of the metal. Since the electron configuration of a lead atom will be directly governed by its speciation (e.g., Pb bound to organics, Pb adsorbed to oxide surfaces, PbS, etc.), X-ray absorption experiments are a powerful in-situ technique for determining speciation that does not suffer from some of the problems of chemical extractions (Bargar et al., 1997a; Bargar et al., 1997b; Bargar et al., 1998).

Selective chemical extractions and synchrotron-based X-ray studies have shown that industrial Pb can be strongly sequestered by organic matter and secondary minerals such as clays and oxides of Al, Fe, and Mn (Miller and McFee, 1983; Jersak et al., 1997; Johnson and Petras, 1998; Kaste et al., 2005). More recent X-ray studies have demonstrated the importance of biomineralization of Pb in soils by bacteria and nematodes (Jackson et al., 2005; Templeton et al., 2003a,b; Xia et al., 1997).

Lead Solid-Solution Partitioning

The concentration of Pb species dissolved in soil solution is probably controlled by some combination of a) Pb mineral solubility equilibria, b) adsorption reactions of dissolved Pb phases on inorganic surfaces (e.g., oxides of Al, Fe, Si, Mn, etc., clay minerals), and c) adsorption reactions of dissolved Pb phases on soil organic matter. Dissolved Pb phases in soil solution can be some combination of Pb^{2+} and its hydrolysis species, Pb bound to dissolved organic matter, and Pb complexes with inorganic ligands such as Cl^- and SO_4^{2-}. Alkaline soils typically have solutions supersaturated with respect to $PbCO_3$, $Pb_3(CO_3)_2(OH)_2$, $Pb(OH)_2$, $Pb_3(PO_4)_2$, $Pb_5(PO_4)_3(OH)$, and $Pb_4O(PO_4)_2$ (Badawy et al., 2002). Pb phosphate minerals in particular, are very insoluble, and thermodynamic data predict that these phases will control dissolved Pb in

soil solution under a variety of conditions (Nriagu, 1974; Ruby et al., 1994). However, certain chelating agents, such as dissolved organic matter can prevent the precipitation of Pb minerals (Lang and Kaupenjohann, 2003).

Soil solution dissolved organic matter content and pH typically have a very strong positive and negative correlation, respectively, with the concentration of dissolved Pb species (Badawy et al., 2002; Sauvé et al., 1998, 2000a,b, 2003; Tipping et al., 2003; Weng et al., 2002). In the case of adsorption phenomena, the partitioning of Pb^{2+} to the solid phase is also controlled by total metal loading: high Pb loadings will result in a lower fraction partitioned to the solid phase. Sauvé et al. (1998; 1997) demonstrated that only a fraction of the total Pb in solution was actually Pb^{2+} in soils treated with leaf compost. The fraction of Pb^{2+} to total dissolved Pb ranged from <1 to 60%, depending on pH and the availability of Pb-binding ligands. In acidic soils, Al species can compete for sites on natural organic matter and inhibit Pb binding to surfaces (Gustafsson et al., 2003).

Tracing the Fate of Atmospherically-delivered Lead

Radiogenic Pb isotopes offer a powerful tool for separating anthropogenic Pb from natural Pb derived from mineral weathering (Erel and Patterson, 1994; Erel et al., 1997). This is particularly useful for studying the mineral soil, where geogenic Pb often dominates. The ore bodies from which anthropogenic Pb are typically derived are usually enriched in ^{207}Pb relative to ^{206}Pb and ^{208}Pb when compared with Pb found in granitic rocks. Uranium-238 series ^{210}Pb also provides a tool for tracing atmospherically-delivered Pb in soils. Fallout ^{210}Pb is deposited onto forests via wet and dry deposition, similar to anthropogenic Pb deposition in forests, and is thus useful as a tracer for non-native Pb in soils. ^{210}Pb is convenient to use for calculating the residence time of Pb in soil layers because it's atmospheric and soil fluxes can be assumed to be in steady-state at undisturbed sites (Dörr, 1995; Dörr and Munnich, 1989; Kaste et al., 2003).

Researchers assessing the fate of atmospheric Pb in soils have also relied on repeated sampling of soils and vegetation for total Pb. This technique works best when anthropogenic Pb accounts for the vast majority of total Pb in a particular reservoir. Johnson et al. (1995b), Yanai et al. (2004), and Friedland et al. (1992) used O horizon (forest floor) time series data to evaluate the movement of gasoline-derived Pb in the soil profile. Surface soils sampled relatively recently demonstrate that the upper soil horizons (O + A horizons) are retaining most of the

anthropogenic Pb burden introduced to the systems during the 20th century (Evans et al., 2005). Miller and Friedland (1994) and Wang and Benoit (1997) suggested that the movement of organic particulates dominated Pb transport in the soil profile.

8.1.1.3 Species Response/Mode of Action

The current document expands upon and updates knowledge since 1986 related to the uptake, detoxification, physiological effects, and modifying factors of lead toxicity to terrestrial organisms. Terrestrial organisms discussed in this chapter include soil organisms, plants, birds, and mammals.

Uptake into Plants and Invertebrates

Recent work supports previous results and conclusions that surface deposition of lead onto above-ground vegetation from airborne sources may be significant (Dalenberg and Van Driel, 1990; Jones and Johnston, 1991; Angelova et al., 2004). In addition, most lead is taken up by plants via the symplastic route (through cell membranes) (Sieghardt, 1990) and remains in the roots, with little translocation to shoots, leaves, or other plant parts. Different species of plants and invertebrates accumulate different amounts of lead (Pižl and Josens, 1995; Terhivuo et al., 1994; Wierzbicka, 1999).

Recent work supports previous conclusions that the form of metal tested, and its speciation in soil, influence uptake and toxicity to plants and invertebrates. The oxide form is less toxic than the chloride or acetate forms, which are less toxic than the nitrate form of lead (Khan and Frankland, 1983; Lock and Janssen, 2002; Bongers et al., 2004). However, these results must be interpreted with caution, as the counterion (e.g., the nitrate ion) may be contributing to the observed toxicity (Bongers et al., 2004).

Detoxification in Plants and Invertebrates

Lead may be deposited in root cell walls as a detoxification mechanism, and this may be influenced by calcium (Antosiewicz, 2005). Yang et al. (2000) suggested that the oxalate content in root and root exudates reduced the bioavailability of lead in soil, and that this was an important tolerance mechanism. Other hypotheses put forward recently include the presence of

sulfur ligands (Sharma et al., 2004) and the sequestration of lead in old leaves (Szarek-Lukaszewska et al., 2004) as detoxification mechanisms.

Lead detoxification has not been studied extensively in invertebrates. Glutathione detoxification enzymes were measured in two species of spider (Wilczek et al., 2004). Lead may be stored in waste nodules in earthworms (Hopkin, 1989) or as pyromorphite in the nematode (Jackson et al., 2005).

Physiological Effects

The effects on heme synthesis (as measured by 5-aminolaevulinic acid dehydratase [ALAD] activity and protoporphyrin concentration, primarily) had been well-documented in the 1986 criteria document and continue to be studied (Schlick et al., 1983; Scheuhammer, 1989; Redig et al., 1991; Henny et al., 1991; Beyer et al., 2000; Hoffman et al., 2000a, b). However, Henny et al. (1991) caution that changes in ALAD and other enzyme parameters are not always related to adverse effects, but simply indicate exposure. Other effects on plasma enzymes, which may damage other organs, have been reported (Brar et al., 1997a, b). Lead also may cause lipid peroxidation (Mateo and Hoffman, 2001) which may be alleviated by Vitamin E, although lead poisoning may still result (Mateo et al., 2003b). Changes in fatty acid production have been reported, which may influence immune response and bone formation (Mateo et al., 2003a).

Response Modification

Genetics, biological factors, physical/environmental factors, nutritional factors and other pollutants can modify terrestrial organism response to lead. Fisher 344 rats were found to be more sensitive to lead than Sprague-Dawley rats (Dearth et al., 2004). Younger animals are more sensitive than older animals (Eisler, 1988; Scheuhammer, 1991), and females generally are more sensitive than males (Scheuhammer, 1987; Tejedor and Gonzalez, 1992; Snoeijs et al., 2005). Monogastric animals are more sensitive than ruminants (Humphreys, 1991). Insectivorous mammals may be more exposed to lead than herbivores (Beyer et al., 1985; Sample et al., 1998), and higher tropic-level consumers may be less exposed than lower trophic-level organisms (Henny et al., 1991). Nutritionally-deficient diets (including low calcium) cause increased uptake of lead (Snoeijs et al., 2005) and greater toxicity (Douglas-Stroebel et al., 2005) in birds.

Mycorrhizal fungi may ameliorate lead toxicity until a threshold is surpassed (Malcová and Gryndler, 2003), which may explain why some studies show increased uptake into plants (Lin et al., 2004) while others show no difference or less uptake (Dixon, 1988). Lower soil pH generally increases uptake of lead into plants and soil invertebrates. However, calcium content, organic matter content, and cation exchange capacity of soils also had a significant influence on uptake of lead into plants and invertebrates (Beyer et al., 1987; Morgan and Morgan, 1988).

Interactions of lead with other metals are inconsistent, depending on the endpoint measured, the tissue analyzed, the animal species, and the metal combination (Phillips et al., 2003; An et al., 2004; Garcia and Corredor, 2004; He et al., 2004; Perottoni et al., 2005).

8.1.1.4 Exposure/Response of Terrestrial Species

The current document expands upon and updates knowledge related to the effects of lead on terrestrial primary producers, consumers and decomposers.

Primary Producers

Effects of lead on terrestrial plants include decreased photosynthetic and transpiration rates, and decreased growth and yield. The phytotoxicity of lead is considered to be relatively low, and there are few reports of phytotoxicity from lead exposure under field conditions. Phytotoxicity data recently were reviewed for the development of the ecological soil screening levels (Eco-SSL) (U.S. Environmental Protection Agency, 2005b). Many of the toxicity data presented in EPA (2005b) are lower than those discussed in the 1986 AQC document, although both documents acknowledge that toxicity is observed over a wide range of concentrations of lead in soil (tens to thousands of mg/kg soil). This may be due to many factors, such as the soil conditions (e.g., pH, organic matter) and differences in bioavailability of the lead in spiked soils, perhaps due to lack of equilibration of the lead solution with the soil after spiking. Most phytotoxicity data continue to be developed for agricultural plant species (i.e., vegetable and grain crops). Few data are available for trees or native herbaceous plants, although two of the five ecotoxicological endpoints used to develop the Eco-SSL were for trees and two were for clover.

Consumers

Effects of lead on avian and mammalian consumers include decreased survival, reproduction, and growth, as well as effects on development and behavior. There remain few field effects data for consumers, except from sites with multiple contaminants, for which it is difficult to attribute toxicity specifically to lead. Avian and mammalian toxicity data recently were reviewed for the development of Eco-SSLs (U.S. Environmental Protection Agency, 2005b). Many of the toxicity data presented in EPA (2005b) are lower than those discussed in the 1986 AQC document, although EPA (2005b) recognizes that toxicity is observed over a wide range of doses (<1 to >1,000 mg Pb/kg bw-day). Most toxicity data for birds are derived from chicken and quail studies, and most data for mammals are derived from laboratory rat and mouse studies. Data derived for other species would contribute to the understanding of lead toxicity, particularly for wildlife species with different gut physiologies. In addition, data derived using environmentally-realistic exposures, such as from lead-contaminated soil and food may be recommended. Finally, data derived from inhalation exposures, which evaluate endpoints such as survival, growth, and reproduction, would contribute to understanding the implications of airborne releases of lead.

Decomposers

Effects of lead on soil invertebrates include decreased survival, growth and reproduction. Effects on microorganisms include changes in nitrogen mineralization, and changes in enzyme activities. Recent data on lead toxicity to soil invertebrates and microorganisms are consistent with those reported in EPA (1986a), with toxicity generally observed at concentrations of hundreds to thousands of mg/kg soil. Studies on microbial processes may be influenced significantly by soil parameters and the significance of the test results is not clear.

Ecological Soil Screening Levels (Eco-SSLs)

Eco-SSLs are concentrations of contaminants in soils that are protective of ecological receptors (U.S. Environmental Protection Agency, 2005a). They were developed following rigorous scientific protocols, and were subjected to two rounds of peer review. The Eco-SSLs for terrestrial plants, birds, mammals, and soil invertebrates are 120 mg/kg, 11 mg/kg, 56 mg/kg and 1700 mg/kg, respectively. See Section 8.1.5 for additional information.

8.1.1.5 Effects of Lead on Natural Terrestrial Ecosystems

At present, industrial point sources such as smelter sites represent the greatest Pb-related threat to the maintenance of sustainable, healthy, diverse, and high-functioning terrestrial ecosystems in the United States. However, assessing the risks specifically associated with Pb is impossible because these sites also experience elevated concentrations of other metals and impacts related to SO_2 emissions. Terrestrial ecosystems may respond to stress in a variety of ways, including reductions in the vigor and/or growth of vegetation, reductions in biodiversity, and effects on microbial processes.

Influence of Dissolved Organic Matter (DOM)

Since the movement and fate of Pb in terrestrial ecosystems is strongly related to the organic matter cycle, stresses that could lead to disruption or alteration of the soil organic matter pool are of particular concern in the assessment of effects of ecosystem stress on Pb cycling. By binding soluble Pb, soil organic matter acts as a barrier to the release of lead to drainage waters (Wang et al., 1995; Kaste et al., 2003; Watmough and Hutchinson, 2004). The release of soluble Pb does not typically result in elevated surface water Pb concentrations because: (1) organic matter has a relatively long residence time in most temperate soils (Gosz et al., 1976; Schlesinger, 1997), so the fraction of the organic matter pool that is dissolved at any time is small; (2) DOM-Pb complexes solubilized in upper soil horizons may be precipitated or adsorbed lower in the soil profile; and, (3) the DOM to which Pb is bound may be utilized by microbes, allowing the associated Pb to bind anew to soil organic matter. These factors moderate the release of Pb to surface waters in temperate terrestrial ecosystems. As a result, concentrations of Pb in soil solutions and drainage waters tend to be low (Driscoll et al., 1988; Bacon and Bain, 1995; Johnson et al., 1995b; Wang et al., 1995). However, stresses or disturbances that result in increased release of DOM from soils could potentially result in the unanticipated release of Pb to groundwater and/or surface waters.

Influence of pH

Like most metals, the solubility of Pb is increased at lower pH (Stumm and Morgan, 1995), suggesting that acidification should result in enhanced mobility of Pb in ecosystems. However, reductions in pH may also cause a decrease in the solubility of DOM, due to the

protonation of carboxylic functional groups (Tipping and Woof, 1990). Because of the importance of complexation with organic matter to Pb mobility in soils, lower DOM concentrations resulting from acidification may offset the increased solubility of the metal. The increased mobility was only observed in very acid soils, those with pH <4.5 (Blake and Goulding, 2002). Acidification also may enhance Pb export to drainage water in very sandy soils, with limited ability to retain organic matter (Swanson and Johnson, 1980; Turner et al., 1985).

Influence of Forest Harvesting

Observations from clear-cut sites in the United States and Europe indicate that forest harvesting causes little or no mobilization of Pb from forest soils (Fuller et al., 1988; Johnson et al., 1995a). The principal risk associated with forest harvesting is the loss of Pb in particulate form to drainage waters through erosion.

Influence of Climate Change

The potential linkages between climate-related stress and Pb cycling are very poorly understood. Effects related to alterations in organic matter cycling may influence Pb migration. For example, an increase in temperature leading to increased rates of organic matter decomposition could lead to temporary increases in DOM concentrations and smaller steady-state pools of soil organic matter. Either of these factors could result in increased concentrations of Pb in waters draining terrestrial ecosystems.

Influence on Microbial Processes

Recent research has documented significant inhibitory effects of Pb and other metals on the activities of several enzymes believed to be crucial to nitrogen mineralization in soils (Senwo and Tabatabai, 1999; Acosta-Martinez and Tabatabai, 2000; Ekenler and Tabatabai, 2002). This suggests that the inhibitory effect of Pb and other metals is broad-based, and not specific to any particular metabolic pathway. In reducing environments, the rate of denitrification is also depressed by trace metals. Fu and Tabatabai (1989) found that 2.5 μmol g^{-1} of Pb (ca. 500 mg kg^{-1}) was sufficient to cause 0, 27, and 52% decreases in nitrogen reductase activity in three different soils.

Effects Observed Around Industrial Point Sources

Assessing the effects of lead exposure on natural ecosystems is complicated by the fact that lead exposure cannot be decoupled from other factors that could also impact the ecosystem under consideration. Principal among these factors are other trace metals and acidic deposition. Emissions of Pb from smelting and other industrial activities are accompanied by other trace metals (e.g., Zn, Cu, Cd) and sulfur dioxide (SO_2) that may cause toxic effects independently or in concert with Pb.

Natural terrestrial ecosystems near smelters, mines, and other industrial activities have exhibited a variety of effects related to ecosystem structure and function. These effects include decreases in species diversity, changes in floral and faunal community composition, and decreasing vigor of terrestrial vegetation. All of these effects were observed in ecosystems surrounding the Anaconda copper smelter, in southwestern Montana, which operated between 1884 and 1980 (Galbraith et al., 1995; Kapustka et al., 1995). Similar observations were made in the area surrounding Palmerton, Pennsylvania, where two zinc smelters operated between 1898 and 1980 (Jordan, 1975; Sopper, 1989; Storm et al., 1994). Subsequent to the effects on vegetation, wind and erosion may remove litter and humus, leaving bare mineral soil, a nearly sterile environment in which very little energy transfer takes place (Little and Martin, 1972; Galbraith et al., 1995). Metal pollution around a Pb-Zn smelter near Bristol, England has not resulted in the loss of oak woodlands within 3 km of the smelter, despite significant accumulation of Pb, Cd, Cu, and Zn in soils and vegetation (Martin and Bullock, 1994). However, the high metal concentrations have favored the growth of metal-tolerant species in the woodland.

The effects of Pb on terrestrial ecosystems near smelters and other industrial sites decrease downwind from the source. Several studies using the soil Pb burden as an indicator have shown that much of the contamination occurs within a radius of 20-50 km around the emission source (e.g., Miller and McFee, 1983; Martin and Bullock, 1994; Galbraith et al., 1995; Spurgeon and Hopkin, 1996a).

8.1.2 Methodologies Used in Terrestrial Ecosystems Research

8.1.2.1 Introduction

The distribution of Pb throughout the ecosystem, via aerial deposition, has been discussed throughout this document. Its further impacts on soil, sediment, and water provide numerous pathways that may promote unacceptable risk to all levels of biota. Stable isotopes of Pb have been found useful in identifying sources and apportionment to various sources. One of the key factors required to assess this risk is an understanding, and perhaps quantification, of bioavailability. Therefore, the bioavailability of Pb is a key issue to the development of ambient air quality criteria (AAQC). However, the discussion of all methods used in characterizing bioavailability is beyond the scope of this chapter. The following topics are discussed in this chapter.

- Lead Isotopes and Apportionment
- Methodologies to determine Pb speciation
- Lead and the Biotic Ligand Model (BLM)
- In situ methods to reduce Pb bioavailability

8.1.2.2 Lead Isotopes and Apportionment

Determination of the extent of Pb contamination from an individual source(s) and its impact are of primary importance in risk assessment. The identification of exposure pathway(s) is fundamental to the risk analysis and critical in the planning of remediation scenarios.

Although societies have been consuming Pb for nearly 9,000 years, production of Pb in the United States peaked in 1910 and 1972, at approximately 750 and 620 kt/year, respectively (Rabinowitz, 2005). The diversity of potential Pb sources (paint pigments, gasoline additives, solders, ceramics, batteries) and associated production facilities (mining, milling, smelting-refining) make fingerprinting of sources difficult. Therefore, dealing with multiple sources (point and non-point), a reliable and specific fingerprinting technique is required. It has been well established (Sturges and Barrie, 1987; Rabinowitz, 1995) that the stable isotope composition of Pb is ideally suited for this task. Lead isotopic ratio differences often allow multiple sources to be distinguished, with an apportionment of the bulk Pb concentration made to those sources.

Lead has four stable isotopes: ^{204}Pb, ^{206}Pb, ^{207}Pb, and ^{208}Pb in natural abundances of 1.4, 24.1, 22.1, and 52.4%, respectively. The radiogenic ^{206}Pb, ^{207}Pb, and ^{208}Pb are produced by radioactive decay of ^{238}U, ^{235}U, and ^{232}Th, respectively. Thus, the isotopic composition of Pb varies based on the U:Pb and Th:Pb ratios of the original ore's source and age (Faure, 1977). Because of the small fractional mass differences of the Pb isotopes, ordinary chemical and pyrometallurgical reactions will not alter their original composition. Therefore, anthropogenic sources reflect the isotopic composition of the ores from which the Pb originated.

To acquire the Pb isotopes, a sample, generally in aqueous form, is analyzed on an ICP/MS (quadrapole, magnetic sector, or time-of-flight). Studies reviewing the most common analytical and sample preparation procedures include Ghazi and Millette (2004), Townsend et al. (1998), and Encinar et al. (2001a,b). The correction factor for mass discrimination biases are generally made by analyzing the National Institute for Standards and Technology (NIST), SRM 981 and/or spiked ^{203}Tl and ^{205}Tl isotopes (Ketterer et al., 1991; Begley and Sharp, 1997). The overall success of Pb isotope fingerprinting is generally dependent on analysis precision, which in turn depends on the type of mass analyzer used (Table 8-1.2.1).

Table 8-1.2.1. Relative Standard Deviation (RSD) for Lead Isotope Ratios on Selected Mass Spectrometers

RSD	Quadrapole	Double-Focusing	Single-Focusing Magnetic Sector	High-Resolution Magnetic Sector ICP/MS
^{204}Pb:^{206}Pb	0.0031	0.0032	0.00053	0.0011
^{207}Pb:^{206}Pb	0.0032	0.0027	0.00053	0.00048
^{208}Pb:^{206}Pb	0.0026	0.0024	0.00053	0.00046

An extensive database comprising primarily North American Pb sources can be assembled from Doe and Rohrbough (1977), Doe and Stacey (1974), Doe et al. (1968), Heyl et al. (1974), Leach et al. (1998), Stacey et al. (1968), Zartman (1974), Cannon and Pierce (1963), Graney et al. (1996), Unruh et al. (2000), James and Henry (1993), Rabinowitz (2005), and Small (1973).

The use of Pb isotopes to quantitatively apportion source contributions follows the simple mixing rule when only two sources are possible (Faure, 1977). Once multiple sources need to be considered, a unique solution can no longer be calculated (Fry and Sherr, 1984). Phillips and Gregg (2003) have designed a model to give feasible source contributions when multiple sources are likely.

Many studies have demonstrated the usefulness of this technique. Media of all types have been studied: water (Flegal et al., 1989a,b; Erel et al., 1991; Monna et al., 1995), ice (Planchon et al., 2002), dust (Adgate et al., 1998; Sturges et al., 1993), and soil/sediments (Hamelin et al., 1990; Farmer et al., 1996; Bindler et al., 1999; Haack et al., 2004; Rabinowitz and Wetherill, 1972; Rabinowitz, 2005; Ketterer et al., 2001).

8.1.2.3 Speciation in Assessing Lead Bioavailability in the Terrestrial Environment

The National Research Council (NRC) 2002 review on bioavailability defined the "bioavailability processes" in terms of three key processes. One of these processes, contaminant interactions between phases, is more commonly referred to as "speciation." For a given metal or metalloid, the term speciation describes the chemical's ability to interact with its biological or chemical surroundings by characterizing its physicochemical properties that are relevant to bioavailability.

A wide variety of analytical (XRD, EMPA, PIXIE, XPS, XAS, SIMS) and chemical species modeling (SOILCHEM, MINTEQL, REDEQL2, ECOSAT, MINTEQA2, HYDRAQL, PHREEQE, WATEQ4F) tools have been used to characterize a metal's speciation as it is found in various media. Currently, for risk assessment purposes (not considering phytotoxicity), where large sites with numerous media, pathways, and metals must often be characterized in a reasonable time frame, EMPA techniques provide the greatest information on metal speciation. Other techniques such as EXAFS and EXANES show great promise and will be important in solving key mechanistic questions. In the case of phytotoxicity, the speciation of metals by direct measurement or chemical models of pore water chemistry is most valuable. Further work needs to be done in developing analytical tools for the speciation of the methyl-forming metals (Hg, As, Sb, Se, and Sn) in soils and sediments.

Concept

As stated above, for a given metal or metalloid (hereafter also referred to as metal), the term speciation refers to its chemical form or species, including its physicochemical characteristics that are relevant to bioavailability. As a result of the direct impact these factors often have on a metal's bioavailability, the term "bioaccessibility" has been adopted to define those factors.

Speciation Role

The accumulation of metals in the lithosphere is of great concern. Unlike organic compounds, they do not degrade and, thus, have a greater tendency to bioaccumulate. It is now well known that knowledge of the bulk, toxic characteristic leaching procedure (TCLP), or synthetic leaching procedure (SLP) concentrations for any metal is not a controlling factor in understanding a metal's environmental behavior or in developing remedies for its safe management. Although these tests are essential to site characterization and management, they offer no insight into risk assessment. Rather, it is the metal's bioavailability (the proportion of a toxin that passes a physiological membrane [the plasma membrane in plants or the gut wall in animals] and reaches a target receptor [in cytosol or blood]), which plays a significant role in the risk assessment of contaminated media.

The National Research Council (NRC) review (NRC, 2002) on bioavailability defined bioavailability processes in terms of three key processes:

- contaminant interactions between phases (association-dissociation/bound-released),
- transport of contaminants to organism, and
- passage across a physiological membrane.

This first process, contaminant interactions between phases, is more commonly referred to as speciation. As described above, the speciation of a toxic metal in the environment is a critical component of any ecosystem health risk assessment. The concept of speciation describes a chemical's ability to interact with its biological or chemical surroundings by characterizing its physicochemical properties. Four important toxicologic and toxicokinetic determinants relating speciation to bioavailability are the (1) chemical form or species, (2) particle size of the metal form, (3) lability of the chemical form, and (4) source.

Chemical Form of Species

The solid phase in a medium controls the activity of a metal in solution, whether the solution is surface, ground, or pore water or GI fluids, and plays a profound role in metal bioavailability. This is perhaps best illustrated by in vivo and in vitro results for many of the common Pb-bearing minerals (Drexler, 1997) (Figure 8-1.2.1). The metal species found in media are often diverse, and data suggest that their bioavailability may be significantly influenced by site-specific variations within these identified metal species (Davis et al., 1993; Ruby et al., 1992; Drexler and Mushak, 1995).

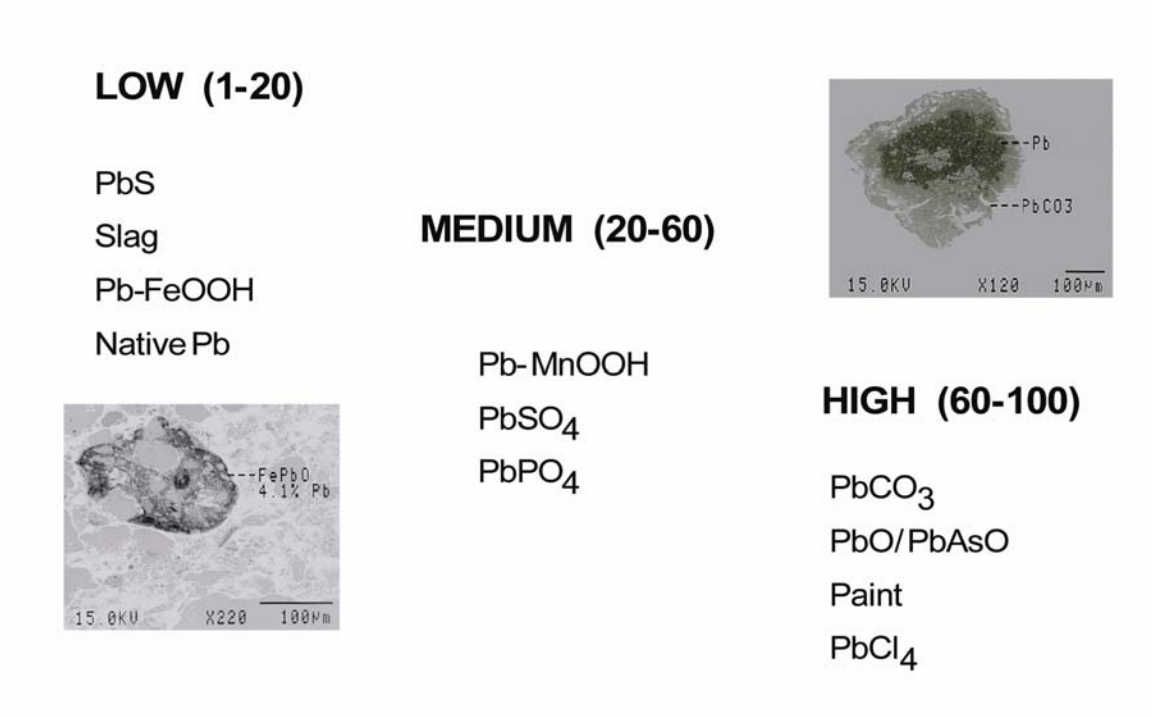

Figure 8-1.2.1. Relationship of bioaccessibility versus speciation.

Particle Size of Metal Species

Particle size of a metal form is an important factor in the mobilization of the metal, primarily because as size decreases, the surface area of the particle increases, thereby increasing solubility. Thus, although solubility is not the only control for bioavailability, an increase in bioavailability has been directly attributed to a decrease in particle size: Barltrop and Meek (1979) observed that "the smaller the lead particle, the higher blood lead level." Similar

observations were made by Healy et al. (1992) using galena (PbS) and an in vitro dissolution technique. Drexler (1997) presented in vitro results on numerous Pb-bearing phases ranging in particle size from 35 to 250 µm. While all phases studied showed increased bioavailability with decreasing particle size, more significantly, not all forms showed the same degree or magnitude of change (Figure 8-1.2.2). Finally, such laboratory data have been supported by extensive epidemiologic evidence, enforcing the importance of particle size (Bornschein et al., 1987; Brunekreef et al., 1983; Angle et al., 1984).

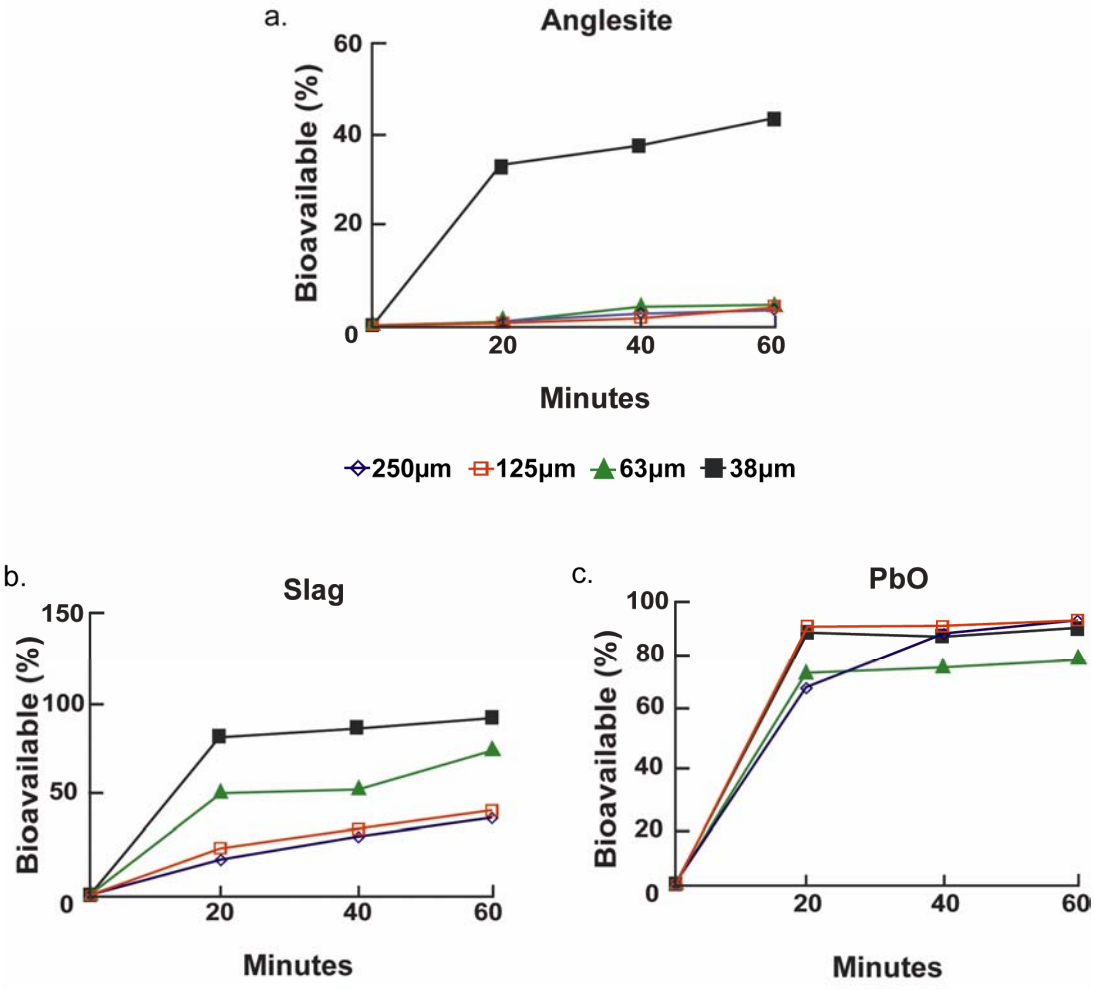

Figure 8-1.2.2. Variation of bioavailability with particle size.

Particle Lability

The impact on bioavailability of a metal particle's lability (its associations within the medium matrix) is not well documented, but it follows the premise put forth by many of the developing treatment technologies regarding its being bound or isolated from its environment. Data from several EPA Superfund sites and the Region VIII swine study (U.S. Environmental Protection Agency, 2004b) suggest that matrix associations, such as liberated versus enclosed, can play an important part in bioavailability. As illustrated in Figure 8-1.2.3, two different media with similar total Pb concentrations and Pb forms (slag, Pb-oxide, and Pb-arsenate) exhibit significantly different bioavailabilities. In the Murray, UT sample (bioaccumulation factor [BAF] = 53%), a greater fraction of the more bioavailable Pb-oxides are liberated and not enclosed in the less-soluble glass-like slag as observed in the Leadville, CO sample (BAF = 17%). Other evidence is more empirical, as illustrated in Figure 8-1.2.4, where a large particle of native Pb is shown to have developed a weathering ring of highly bioavailable Pb-chloride and Pb-oxide. Such observations can be useful in understanding the mechanistic phenomena controlling bioavailability. In addition, they will aid in developing and validating models to predict metal-environment interactions.

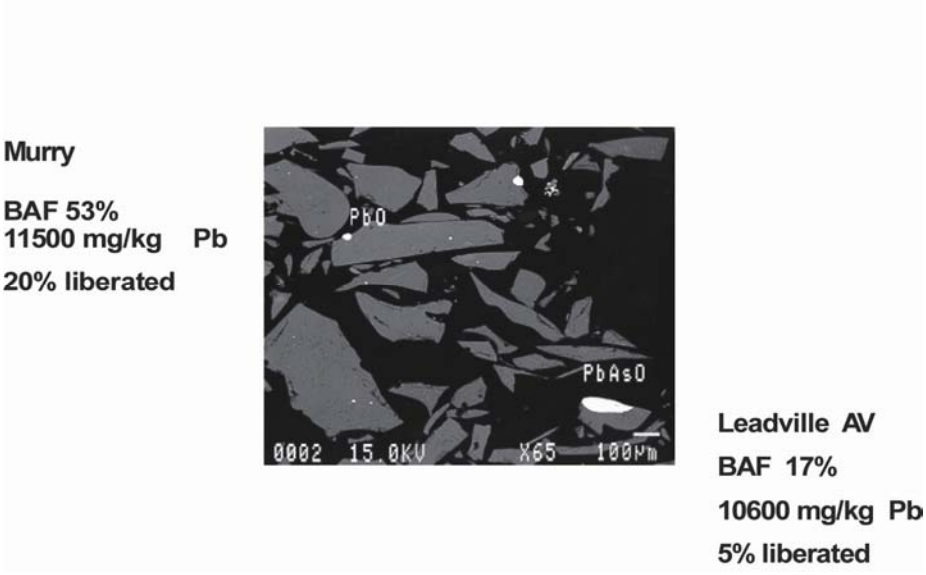

Figure 8-1.2.3. Illustration of particle lability and bioavailability.

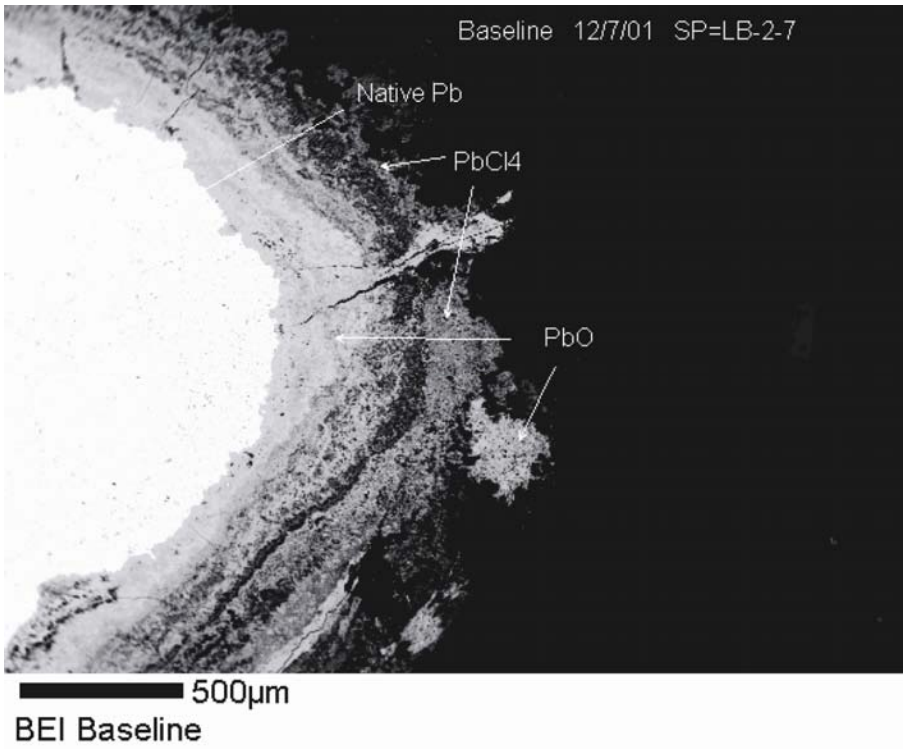

Figure 8-1.2.4. Scanning electron micrograph of a large native Pb particle.

Source

Although the source of a metal is not directly related to bioavailability, it plays an important role in risk assessment with the evaluation of metal (1) pathways, (2) background, and (3) apportionment. It is important to understand a metal's pathway before any remedial action can be taken; otherwise, recontamination of the primary pathway and reexposure can occur. Knowledge of background is required by statute, as an action level cannot be established below natural background levels. Finally, cost recovery can be an important factor in a remedial action, as it is the Agency's responsibility to identify and, if possible, seek the assistance of responsible parties.

Plants

When considering the bioavailability of a metal to plants from soils and sediments, it is generally assumed that both the kinetic rate of supply and the speciation of the metal to either the root or shoot are highly important. In soils and sediments, generally only a small volume of

water is in contact with the chemical form, and although the proportion of a metal's concentration in this pore water to the bulk soil/sediment concentration is small, it is this phase that is directly available to plants. Therefore, pore water chemistry (i.e., metal concentration as simple inorganic species, organic complexes, or colloid complexes) is most important.

Tools currently used for metal speciation for plants include (1) in situ measurements using selective electrodes (Gundersen et al., 1992; Archer et al., 1989; Wehrli et al., 1994); (2) in situ collection techniques using diffusive equilibrium thin films (DET) and diffusive gradient thin films (DGT) followed by laboratory analyses (Davison et al., 1991, 1994; Davison and Zhang, 1994; Zhang et al., 1995); and (3) equilibrium models (SOILCHEM) (Sposito and Coves, 1988).

8.1.2.4 Tools for Bulk Lead Quantification and Speciation

Bulk Quantification

The major analytical methods most commonly used for bulk analyses outlined in the 1986 Pb ACQD included:

- Atomic Absorption Spectrometry (AAS)
- Emission Spectrometry (Inductively coupled plasma/atomic emission spectrometry)
- X-ray Fluorescence (XRF)
- Isotope Dilution Mass Spectrometry (ID/MS)
- Colorimetric
- Electrochemical (anodic stripping voltametry and differential pulse polarography).

The choice of analytical method today for bulk quantification is generally ICP/AES or ICP/MS (U.S. Environmental Protection Agency, 2001). Since 1986, numerous standard reference materials (SRM) have been developed for Pb (Table 8-1.2.2), and several significant technological improvements have been developed.

Modern spectrometry systems have replaced photomultiplier tubes with a charge-coupled device (CCD). The CCD is a camera that can detect the entire light spectrum (>70,000 lines) from 160 to 785 nm. This allows for the simultaneous measurement of all elements, as well as any interfering lines (a productivity increase), and increases precision. The detection limit for Pb in clean samples can now be as low as 40 ppb.

Table 8-1.2.2. National Institute of Standards and Technology Lead SRMs

NIST SRM	Medium	Mean Pb mg/kg
2710	Soil	5532
2711	Soil	1162
2709	Soil	18.9
2587	Soil (paint)	3242
2586	Soil (paint)	432
2783	Filter ($PM_{2.5}$)	317
1648	Urban particulate	6550
1649a	Urban dust	12,400
2584	Indoor dust	9761
2583	Indoor dust	85.9
1515	Apple leaves	0.47
1575	Pine needles	0.167

Modern ICP/AES systems offer a choice of either axial viewed plasma (horizontal), which provides greater sensitivity (DL= 0.8 µg/L Pb), or radial (vertical) viewed plasma, which performs best with high total dissolved samples (DL = 5.0 µg/L Pb).

The development of reaction or collision cells have expanded the capabilities of ICP/MS and lowered detection limits for many elements that were difficult to analyze because of interferences such as Se, As, Ti, Zn, Ca, Fe, and Cr. The cells provide efficient interference (isobaric, polyatomic, and argide) removal independent of the analyte and sample matrix by using various reaction gases (H_2, He, NH_3), eliminating the need for interference correction equations.

Speciation Tools

A wide variety of analytical and chemical techniques have been used to characterize a metal's speciation (as defined above) in various media (Hunt et al., 1992; Manceau et al., 1996,

2000a; Welter et al., 1999; Szulczewski et al., 1997; Isaure et. al., 2002; Lumsdon and Evans, 1995; Gupta and Chen, 1975; Ma and Uren, 1995; Charlatchka et al., 1997). Perhaps the most important factor that one must keep in mind in selecting a technique is that, when dealing with metal-contaminated media, one is most often looking for the proverbial "needle in a haystack." Therefore, the speciation technique must not only provide the information outlined above, but it must also determine that information from a medium that contains very little of the metal. As illustrated in Figure 8-1.2.5, for a Pb-contaminated soil, less than 1% (modally) of a single species can be responsible for a bulk metals concentration above an action level. This factor is even more significant for other metals (i.e., As, Cd, or Hg) were action levels are often below 100 mg/kg.

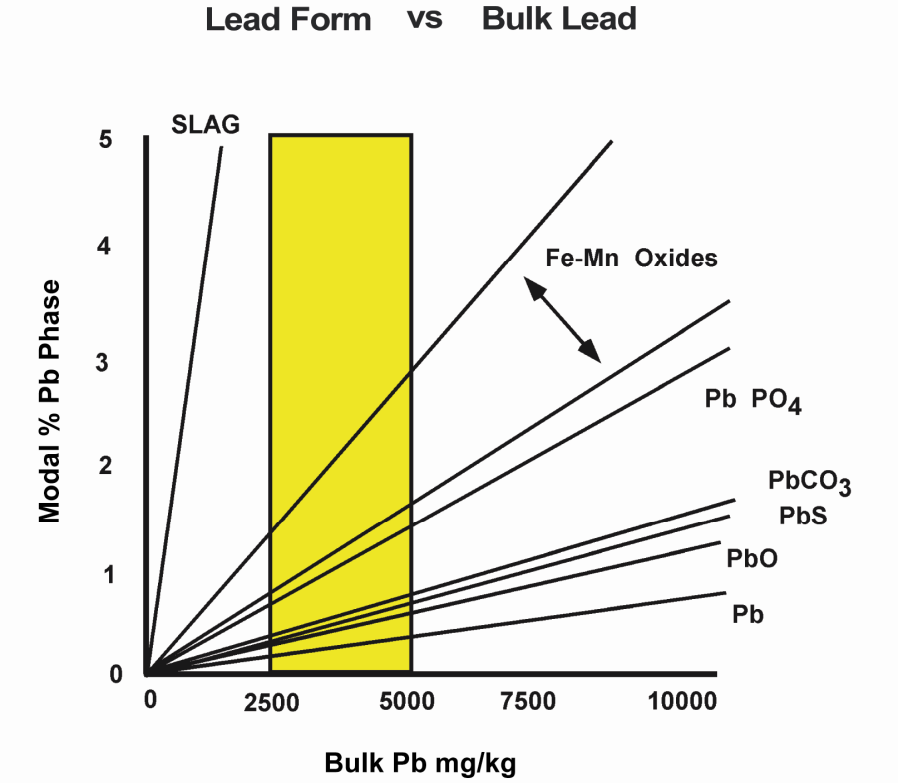

Figure 8-1.2.5. Bulk lead versus single species modality.

Of the techniques tested (physicochemical, extractive, and theoretical), the tools that have been used most often to evaluate speciation for particle-bound metal include X-ray absorption

spectroscopy (XAS), X-ray diffraction (XRD), particle induced X-ray emission (PIXE and μPIXE), electron probe microanalysis (EPMA), secondary ion mass spectrometry (SIMS), X-ray photoelectron spectroscopy (XPS), sequential extractions, and single chemical extractions. The tools that have been used most often to evaluate speciation for metal particles in solution include the following computer-based models: MINTEQL, REDEQL2, ECOSAT, MINTEQA2, HYDRAQL, PHREEQE, and WATEQ4F. These tools are briefly described below.

Particle-Bound Metal

Direct Approaches

Over the past decade, numerous advances in materials science have led to the development of a wide range in analytical tools for the determination of metal concentration, bonding, and valance of individual particles on a scale that can be considered useful for the speciation of environmentally important materials (i.e., soils, wastes, sediments, and dust). This review will provide the reader with a brief description of these techniques, including their benefits, limitations (cost, availability, sample preparation, resolution), and usability as well as references to current applications. Although most of these tools are scientifically sound and offer important information on the mechanistic understanding of metal occurrence and behavior, only a few currently provide useful information on metal bioavailability at a "site" level. However, one may still find other techniques essential to a detailed characterization of a selected material to describe the chemical or kinetic factors controlling a metal's release, transport, and/or exposure.

X-Ray absorption Spectroscopy (XAS). X-ray absorption spectroscopy (XAS) is a powerful technique using the tunable, monochromatic (white light) X-rays produced by a synchrotron (2-4 GeV) to record oscillations in atomic absorption within a few 100 eV of an element's absorption edge. Spectra provide both information on chemical state and atomic structure. Measurements are theoretically available for all elements and are not surface-sensitive nor sample-sensitive (i.e., gases, liquids, solids, and amorphous materials are testable).

High-energy spectra within 30 eV of the edge are termed XANES (X-ray absorption near edge structure spectroscopy (Fendorf et al., 1994; Maginn, 1998) are particularly suited for determination and quantification (10 to 100 ppm) of metal in a particular oxidation state

(Szulczewski et al., 1997; Shaffer et al., 2001; Dupont et al., 2002). The lower-energy spectra persist some 100 eV above the edge. These oscillations are termed EXAFS (extended X-ray absorption fine structure) and are more commonly used for speciation analyses (Welter et al., 1999; Manceau et al., 1996, 2000a; Isaure et al., 2002).

The main limitations to XAS techniques are (1) the lack of spatial resolution; (2) XAS techniques provide only a weighted average signal of structural configurations, providing information on the predominant form of the metal, while minor species, which may be more bioavailable, can be overlooked; (3) access to synchrotrons is limited and the beam time required to conduct a site investigation would be prohibitive; (4) a large spectral library must be developed; (5) generally, poor fits to solution models are achieved when the compound list is large; and (6) high atomic number elements have masking problems based on compound density.

X-Ray Diffraction (XRD). In X-ray diffraction, a monochromatic Fe, Mo, Cr, Co, W, or Cu X-ray beam rotates about a finely powdered sample and is reflected off the interplanar spacings of all crystalline compounds in the sample, fulfilling Bragg's law ($n\lambda = 2d\sin\theta$). The identification of a species from this pattern is based upon the position of the lines (in terms of θ or 2θ) and their intensities as recorded by an X-ray detector. The diffraction angle (2θ) is determined by the spacing between a particular set of atomic planes. Identification of the species is empirical, and current available databases contain more than 53,000 compounds.

If a sample contains multiple compounds, interpretation becomes more difficult and computer-matching programs are essential. In some instances, by measuring the intensity of the diffraction lines and comparing them to standards, it is possible to quantitatively analyze crystalline mixtures; however, if the species is a hydrated form or has a preferred orientation, this method is only semiquantitative at best. Since this technique represents a bulk analysis, no particle size or lability information can be extracted from the patterns.

Particle Induced X-Ray Emission (PIXE and μPIXE). Particle induced X-ray emission (PIXE) uses a beam, ~4 μm in diameter, of heavy charged particles (generally He) to irradiate the sample. The resulting characteristic X-rays are emitted and detected in a similar manner as XRF, using Si-Li detectors. Particles generated from a small accelerator or cyclotron, with a potential of 2 to 4 MeV, are commonly used. Detection limits on the order of 1 mg/kg are achieved on thin-film samples. Disadvantages to its use for speciation include (1) only a small volume of material can be analyzed (1 to 2 mg/cm^2); (2) no particle size information is provided;

(3) peak overlaps associated with Si-Li detectors limit identification of species; (4) limited availability; and (5) high cost. For a further review of PIXE analysis and applications, see Maenhaut (1987).

Electron Probe Microanalysis (EPMA). Electron probe microanalysis uses a finely focused (1 μm) electron beam (generated by an electron gun operating at a 2 to 30 Kv accelerating voltage and pico/nanoamp currents) to produce a combination of characteristic X-rays for elemental quantification along with secondary electrons and/or backscatter electrons for visual inspection of a sample. Elements from beryllium to uranium can be nondestructively analyzed at the 50 ppm level with limited sample preparation. X-ray spectra can be rapidly acquired using either wavelength dispersive spectrometers (WDS) or energy dispersive spectrometers (EDS).

With WDS, a set of diffracting crystals, of known d-spacing, revolve in tandem with a gas-filled proportional counter inside the spectrometer housing so that Bragg's law is satisfied and a particular wavelength can be focused. Photon energy pulses reflecting off the crystal are collected for an individual elemental line by the counter as a first approximation to concentration. For quantitative analysis, these intensities are compared to those of known standards and must be corrected for background, dead time, and elemental interactions (ZAF) (Goldstein et al., 1992). ZAF correction is in reference to the three components of matrix effects: atomic number (Z), absorption (A), and fluorescence (F).

With EDS, a single Si-Li crystal detector is used in conjunction with a multichannel analog-digital converter (ADC) to sort electrical pulses (with heights approximately proportional to the quantum energy of the photon that generated them), producing a spectrum of energy (wavelength) versus counts. The net area under a particular peak (elemental line) is proportional to its concentration in the sample. For quantitative analyses, corrections similar to WDS analysis must be performed. Although EDS detectors are more efficient than WDS, detection limits are significantly greater (~1000 ppm), because of elevated backgrounds and peak overlaps. For speciation analysis, the EDS system must NEVER be used as the primary detector, as numerous errors in species identification are often made. These are generally the result of higher-order X-ray line overlaps.

This technique has been routinely used for site characterizations (Linton et al., 1980; Hunt et al., 1992; Camp, Dresser, and McKee (CDM), 1994; U.S. Environmental Protection

Agency, 2002). Currently this technique offers the most complete data package on metal speciation than any of the other tools. The method is relatively fast and inexpensive, available, and provides all of the required information for bioavailability assessments (i.e., particle size, species, lability, and sourcing). A number of limitations still need to be addressed including: (1) its inability to quickly isolate a statistically significant population of particles in soils with low bulk metal concentrations (<50 mg/kg), meaning that for some metals with low concentrations of concern (i.e., Cd, Mo, Sb, Se), this method may be less useful; (2) the more volatile metals (i.e., Hg, Tl) are often volatilized under the electron beam or lost during sample preparation.

Secondary Ion Mass Spectrometry (SIMS). Secondary ion mass spectrometry (also known as ion microprobes or ion probes) is a well-known primarily surface technique that uses a 0.5 to 20 KV O, Ar, Ga, In, or Cs ion beam in bombarding (sputtering) the surface of a sample while emitting secondary ions that are detected by either quadrapole, time-of-flight (TOF), or magnetic sector mass spectrometers. Sensitivity is very high, in the ppb range for elements hydrogen to uranium, providing quantitative results on elemental or isotopic metals and organic compounds. With the advent of liquid metal (In and Ga) ion beams, beam sizes of less than 1 μm are possible, although 20 μm is more commonly used.

The major disadvantage of SIMS to species identification is that each element or isotope must be tuned and analyzed sequentially. This makes the identification of a metal form highly time-consuming and, thus, the characterization of a multiphase medium impractical.

X-Ray Photoelectron Spectroscopy (XPS). X-ray photoelectron spectroscopy (XPS) or ESCA (electron spectroscopy for chemical analysis, as it was previously known) is a classical surface, 10 to 50 Å in depth, analytical technique for determinating qualitative elemental concentrations of elements greater than He in atomic number and provides limited structural and oxidation state information. In XPS, the high-energy (15 Kv) electrons are typically produced from a dual-anode (Al-Mg) X-ray tube. The excitation or photoionization of atoms within the near surface of the specimen emit a spectrum of photoelectrons. The measured binding energy is characteristic of the individual atom to which it was bound. Monochromatic sources are often employed to improve energy resolution, allowing one to infer oxidation states of elements or structure of compounds (organic and inorganic) by means of small chemical shifts in binding energies (Hercules, 1970). The major disadvantages of XPS for environmental speciation studies

is its poor sensitivity, especially in complex matrices and its large, 100-200 μm, spatial resolution. Direct speciation techniques discussed above are summarized in Table 8-1.2.3.

Indirect Approaches

A more indirect approach to speciation than the methods previously described include the functional or operational extraction techniques that have been used extensively over the years (Tessier et al, 1979; Tessier and Campbell, 1988; Gupta and Chen, 1975). These methods use either a single or sequential extraction procedure to release species associated with a particular metal within the media. Single chemical extractions are generally used to determine the bioavailable amount of metal in a functional class: water-soluble, exchangeable, organically bound, Fe-Mn bound, or insoluble.

In a similar approach, sequential extractions treat a sample with a succession of reagents intended to specifically dissolve different, less available phases. Many of these techniques have been proposed, most of which are a variation on the classical method of Tessier et al. (1979), in which metal associated with exchangeable, carbonate-bound, Fe-Mn bound, organically bound, and residual species can be determined. Beckett (1989), Kheboian and Bauer (1987), and Foerstner (1987) provide excellent reviews on the use and abuse of extractions. These techniques can be useful in a study of metal uptake in plants, where transfer takes place predominately via a solution phase. However, one must keep in mind that they are not "selective" in metal species, give no particle size information and, above all, these leachable fractions have never been correlated to bioavailability.

Solution Speciation Using Computer-Based Models. Computer-based models are either based upon equilibrium constants or upon Gibb's free energy values in determining metal speciation from solution chemistry conditions (concentration, pH, Eh, organic complexes, adsorption/desorption sites, and temperature). Both approaches are subject to mass balance and equilibrium conditions. These models have undergone a great deal of development in recent years, as reliable thermodynamic data has become available and can provide some predictive estimates of metal behavior. A good review of these models and their applications is provided by Lumsdon and Evans (1995).

Table 8-1.2.3. Characteristics for Direct Speciation Techniques

Tools	Species Lability	Species Particle Size	Species Valance State	Species Bonding	Species Composition	Species Abundance	Element Specificity	Isotopic Character	Element Sensitivity	Resolution	Availability	Cost
XRD	No	No	No	No	No#	No	No	No	3-4 vol%	Bulk	1	$
EMPA	Yes	Yes	Yes+	No	Yes	Yes?	B-U	No***	50 ppm	0.5-1 μ	2	$$
SIMS	No	Yes	No	No	Yes*	Yes**	Li-U	Yes	1 ppb	10 μ	4	$$$
XPS	No	No	Yes	Yes	Yes*	Yes**	H-U	No	wt.%	100 μ	2	$$
XAS	No	No	Yes	Yes	Yes*	Yes**	He-U	No	ppb	2 μ	5	$$$$
PIXIE	No	No	No	No	Yes	Yes**	B-U	No	10 ppm	4 μ	4	$$$$

*Technique requires each element be tuned and standardized, requiring unreasonable time limits.
** Techniques designed and tested only on simple systems. Multiple species require lengthy analytical times and data reduction.
*** Limited when combined with ICP/MS/LA.
# Identifies crystalline compounds and stoichiometric compositions only.
? Technique has limitations based on particle counting statistics.
+ Valance determined by charge balance of complete analyses.

Speciation can be controlled by simple reactions; however, in many cases, particularly in contaminated media, their state of equilibrium and reversibility are unknown. In addition, these models suffer from other limitations such as a lack of reliable thermodynamic data on relevant species, inadequacies in models to correct for high ionic strength, reaction kinetics are poorly known, and complex reactions with co-precipitation/adsorption are not modeled.

The first limitation is perhaps the most significant for contaminated media. For example, none of the models would predict the common, anthropogenic, Pb phases, i.e., paint, solder, and slag.

8.1.2.5 Biotic Ligand Model

The Biotic ligand model (BLM) is an equilibrium-based model that has been incorporated into regulatory agencies guidelines (including the EPA) to predict effects of metals on aquatic and terrestrial biota and to aid in the understanding of their interactions with biological surfaces. Most recent directions in research are directed toward extending the model to predict metal toxicity in soils.

As initially presented by Paquin et al. (1999), the BLM evolved from both the gill surface interaction model (GSIM) of Pagenkopf (1983) and the free ion activity model (FIAM) of Morel (1983). The model can be used to define site-specific ambient water quality criteria (AWQC) by providing the rational as to how metal toxicity to an aquatic organism is controlled by variations in water chemistry.

By integrating the interaction of a metal in solution with its predicted speciation and subsequent interaction with either a receptor site (e.g., root, gill, whole body) of an organism (biotic ligand) a lethal concentration (LC_{50}) estimate is made, replacing expensive, time consuming bioassay testing. The biotic ligand is assumed to be independent and homogeneously distributed and is essentially described using an affinity constant (Ks [M-1]) that have been generated from laboratory studies. A current version (v 2.12) of the BLM can be downloaded from: http://www.hydroqual.com/blm.

Currently, a limited metal/organism set ([Cu, Ag, Cd, and Zn] and [flathead minnow, rainbow trout, *Daphnia magna*, *Daphnia pulex*, and *Ceidaphia dubia*], respectively) are provided. However, users are able to input site-specific metal/organism datasets if available. The literature contains numerous studies on additional metals (i.e., Co, Ni, Pb, U, Sr, and Ba)

and aquatic organisms, references to which can be found in Slaveykova and Wilkinson (2005) and Niyogi and Wood (2004). Site-specific water chemistry is entered as temperature, pH, metal (Cu, Ag, Cd, and Zn), dissolved organic carbon (DOM), humic acid (HA), cations (Ca, Mg, Na, and K), anions (Cl and SO_4), and alkalinity for speciation calculations.

Currently, there is no acute BLM for Pb; however the work of MacDonald et al. (2002) on gill-Pb in rainbow trout and that of Slaveykova and Wilkinson (2002) on algae suggest that Ca^{2+}, DOM, and perhaps Na^{+} competitively inhibit Pb^{2+} uptake and thus exhibit a much lower (<100×) affinity for the biotic ligand. Further toxicity testing must be conducted before an acute BLM for Pb is established. Presently, affinity constants for Pb are limited to a few organisms (Table 8-1.2.4).

Table 8-1.2.4. Affinity Constants for Lead

Organism	Species	log Ks [M-1]	Reference
Phytoplankton	*Chlorella kesslerii*	5.5	Slaveykova and Wilkinson (2002)
Bacteria	*Bacillus subtilis*	3.4, 5.1	Daughney and Fein (1998)
	Bacillus lichiformis	4.4, 5.7	Daughney and Fein (1998)
Fish	Rainbow trout	6.0	MacDonald et al. (2002)
Cladoceran	*Hyalella azteca*	5.8, 6.9	Borgmann et al. (1993, 2004) MacLean et al. (1996)

8.1.2.6 Soil Amendments

The removal of contaminated soil to mitigate exposure of terrestrial ecosystem components to Pb can often present both economic and logistic problems. Because of this, recent studies have focused on in situ methodologies to lower soil-Pb RBA (Brown et al., 2003a,b). To date, the most common methods studied include the addition of soil amendments in an effort to either lower the solubility of the Pb form or to provide sorbtion sites for fixation of pore-water Pb. These amendments typically fall within the categories of phosphate, biosolid, and Al/Fe/Mn-oxide amendments.

Phosphate Amendments

Phosphate amendments have been studied extensively and, in some cases, offer the most promising results (Brown et al., 1999; Ryan et al., 2001; Cotter-Howells and Caporn, 1996; Hettiarachchi et al., 2001, 2003; Yang et al., 2001; Ma et al., 1995). Research in this area stems from early work by Nriagu (1973) and Cotter-Howells and Caporn (1996), who pointed out the very low solubilities for many Pb-phosphates (Ksp −27 to −66), particularly chloropyromorphite [$Pb_5(PO_4)_3Cl$]. The quest to transform soluble Pb mineralogical forms into chloropyromorthite continues to be the primary focus of most studies. Sources of phosphorous have included phosphoric acid (H_3PO_4), triple-super phosphate (TSP), phosphate rock, and/or hydroxyapatite (HA). Various studies have combined one or more of these phosphorous sources with or without lime, iron, and/or manganese in an attempt to enhance amendment qualities. Most amendments are formulated to contain between 0.5 and 1.0% phosphorous by weight. They are then either applied wet or dry and then mixed or left unmixed with the contaminated soil. Success of phosphate amendments has been variable, and the degree of success appears to depend on available phosphorous and the dissolution rate of the original Pb species.

A number of potentially significant problems associated with phosphate amendments have been recognized, including both phyto- and earthworm toxicity (Ownby et al., 2005; Cao et al., 2002; and Rusek and Marshall, 2000). Both of these toxicities are primarily associated with very high applications of phosphorous and/or decreased soil pH. Indications of phytotoxicity are often balanced by studies such as Zhu et al. (2004) that illustrate a 50 to 70% reduction in shoot-root uptake of Pb in phosphate-amended soils. Additionally, the added phosphate poses the potential risk of eutrophication of nearby waterways from soil runoff.

Biosolid Amendments

Historically, biosolids have been used in the restoration of coal mines (Haering et al., 2000; Sopper, 1993). More recently, workers have demonstrated the feasibility of their use in the restoration of mine tailings (Brown et al., 2003a), and urban soils (Brown et al., 2003b; Farfel et al., 2005). Mine tailings are inherently difficult to remediate in that they pose numerous obstacles to plant growth. They are most often (1) acidic; (2) high in metal content, thus prone to phytotoxicity; (3) very low in organic content; and (4) deficient in macro- and micronutrients.

Stabilization (i.e., the establishment of a vegetative cover) of these environments is essential to the control of metal exposure or migration from soil/dust and groundwater pathways.

At Bunker Hill, ID, Brown et al. (2003b) demonstrated that a mixture of high nitrogen biosolids and wood pulp or ash, when surface applied at a rate of approximately 50 and 220 tons/ha, respectively, increased soil pH from 6.8 to approximately 8.0, increased plant biomass from 0.01 mg/ha to more than 3.4 tons/ha, and resulted in a healthy plant cover within 2 years. Metal mobility was more difficult to evaluate. Plant concentrations of Zn and Cd were generally normal for the first 2 years of the study; however, Pb concentrations in vegetation dramatically increased two to three times in the first year. Additionally, macronutrients (Ca, K, and Mg) decreased in plant tissue.

Urban soils, whether contaminated from smelting, paint, auto emissions, or industrial activity, are often contaminated with Pb (Agency for Toxic Substances and Disease Registry [ATSDR], 1988) and can be a significant pathway to elevated child blood Pb levels (Angle et al., 1974). Typically, contaminated residential soils are replaced under Superfund rules. However, urban soils are less likely to be remediated unless a particular facility is identified as the contaminate source. Application of biosolids to such soils may be a cost-effective means for individuals or communities to lower Pb RBAs.

A field study by Farfel et al. (2005) using the commercial biosolid ORGO found that, over a 1-year period, Pb in the dripline soils of one residence had reduced RBAs by approximately 60%. However, soils throughout the remainder of the yard showed either no reduction in RBA or a slight increase. A more complex study was conducted by Brown et al. (2003a) on an urban dripline soil in the lab. The study used an assortment of locally derived biosolids (raw, ashed, high-Fe compost, and compost) with and without lime. All amendments were incubated with approximately 10% biosolids for a little more than 30 days. In vitro and in vivo data both indicated a 3 to 54% reduction in Pb RBA, with the high-Fe compost providing the greatest reduction.

As with phosphate amendments, problems with biosolid application have also been documented. Studies have shown that metal transport is significantly accelerated in soils amended with biosolids (Al-Wabel et al., 2002; McBride et al., 1999, 1997; Lamy et al., 1993; Richards et al., 1998, 2000). Some of these studies indicate that metal concentrations in soil solutions up to 80 cm below the amended surface increased by 3- to 20-fold in concentration up

to 15 years after biosolid application. The increase in metal transport is likely the result of elevated dissolved organic carbon (DOC) in the amended soil. Anodic stripping voltammetry has indicated that very low (2 to 18%) of the soluble metals are present as ionic or inorganic complexes (McBride, 1999; Al-Wabel et al., 2002).

8.1.2.7 Future Needs

Since the 1986 Lead ACQD, considerable data has been generated on the bioavailability process. The understanding of bioavailability is central to improving risk assessments and designing efficient, cost-effective remediations. Four key areas for future research can be identified.

- A set of bioavailability and speciation standards should be developed for traceability and quality assurance to aid researchers in developing new or refining existing tools.
- An effort should be made to develop in vitro bioassays for nonhuman biota in order to provide site-specific, rapid, cost-effective estimates of bioavailability/toxicity for all levels of the ecosystem evaluated in a risk assessment.
- Research should continue on the development of in situ amendments to lower Pb bioavailability, with a strong emphasis on long-term field validation studies.
- Finally, toxicity testing for expanding organism/metal affinity constants for the BLM should be continued.

8.1.3 Distribution of Atmospherically Delivered Lead in Terrestrial Ecosystems

8.1.3.1 Introduction

The 1986 Lead Air Quality Criteria Document (Pb AQCD) (U.S. Environmental Protection Agency, 1986a) contains only a few minor sections that detail the speciation, distribution, and behavior of atmospherically delivered Pb in terrestrial ecosystems. The document concluded that the majority of Pb in the atmosphere at that time was from gasoline consumption: of the 34,881 tons of Pb emitted to the atmosphere in 1984, 89% was from gasoline use and minor amounts were from waste oil combustion, iron and steel production, and smelting. Lead in the atmosphere today, however, does not come from gasoline consumption; instead it results largely from waste incineration, metal smelting, and metal production (Polissar et al., 2001; Newhook et al., 2003). The emission source can determine the species of Pb that are

delivered to terrestrial ecosystems. For example, Pb species emitted from automobile exhaust is dominated by particulate Pb halides and double salts with ammonium halides (e.g., PbBrCl, $PbBrCl_2NH_4Cl$), while Pb emitted from smelters is dominated by Pb-sulfur species (Habibi, 1973). The halides from automobile exhaust break down rapidly in the atmosphere, possibly via reactions with atmospheric acids (Biggins and Harrison, 1979). Lead phases in the atmosphere, and presumably the compounds delivered to the surface of the earth (i.e., to vegetation and soils), are suspected to be in the form of $PbSO_4$, PbS, and PbO (Olson and Skogerboe, 1975; Clevenger et al., 1991; Utsunomiya et al., 2004).

There are conflicting reports of how atmospherically derived Pb specifically behaves in surface soils. This disagreement may represent the natural variability of the biogeochemical behavior of Pb in different terrestrial systems, or it may be a function of the different analytical methods employed. The importance of humic and fulvic acids (Zimdahl and Skogerboe, 1977; Gamble et al., 1983) and hydrous Mn- and Fe-oxides (Miller and McFee, 1983) for scavenging Pb in soils are discussed in some detail in the 1986 Pb AQCD. Nriagu (1974) used thermodynamics to argue that Pb-orthophosphates (e.g., pyromorphite) represented the most stable Pb phase in many soils and sediments. He further suggested that, because of the extremely low solubility of Pb-phosphate minerals, Pb deposition could potentially reduce phosphorous availability. Olson and Skogerboe (1975) reported that solid-phase $PbSO_4$ dominated gasoline-derived Pb speciation in surface soils from Colorado, Missouri, and Chicago, while Santillan-Medrano and Jurinak (1975) suggested that $Pb(OH)_2$, $Pb(PO_4)_2$, and $PbCO_3$ could regulate Pb speciation in soils. However, insoluble organic material can bind strongly to Pb and prevent many inorganic phases from ever forming in soils (Zimdahl and Skogerboe, 1977).

The vertical distribution and mobility of atmospheric Pb in soils was poorly documented prior to 1986. Chapter 6 of the 1986 AQCD cited a few references suggesting that the atmospheric Pb is retained in the upper 5 cm of soil (Reaves and Berrow, 1984). Techniques using radiogenic Pb isotopes had been developed to discern between gasoline-derived Pb and natural, geogenic (native) Pb, but these techniques were mostly applied to only sediments (Shirahata et al., 1980) prior to the 1986 Pb AQCD. Without using these techniques, accurate determinations of the depth-distribution and potential migration velocities for atmospherically delivered Pb in soils were largely unavailable.

Some technological advances, combined with the expansion of existing technologies after 1986, resulted in the publication of a large body of literature detailing the speciation, distribution, and geochemical behavior of gasoline-derived Pb in the terrestrial environment. Most notably, the development of selective chemical extraction (SCE) procedures as a rapid and inexpensive means for partitioning Pb into different soil and sediment phases (e.g., Pb-oxides, Pb-humate, etc.) has been exploited by a number of researchers (Tessier et al., 1979; Johnson and Petras, 1998; Ho and Evans, 2000; Scheckel et al., 2003). Also, since 1986, several workers have exploited synchrotron-based XAS in order to probe the electron coordination environment of Pb in soils, organic matter, organisms, and sediments (Manceau et al., 1996; Xia et al., 1997; Trivedi et al., 2003). X-ray absorption studies can be used for the in situ determination of the valence state of Pb and can be used to quantify Pb speciation in a variety of untreated samples. Biosensors, which are a relatively new technology coupling biological material such as an enzyme with a transducer, offer a new, simple, and inexpensive means for quantifying available Pb in ecosystems (Verma and Singh, 2005). Advances in voltammetric, diffusive gradients in thin films (DGT), and ICP techniques have also increased the abilities of researchers to quantify Pb phases in solutions (Berbel et al., 2001; Scally et al., 2003). In addition to the development of techniques for describing and quantifying Pb species in the soils and solutions, researchers have used radiogenic Pb isotopes (^{206}Pb, ^{207}Pb, ^{208}Pb) to quantify the distribution, speciation, and transport of anthropogenic Pb in soil profiles and in vegetation (Bindler et al., 1999; Erel et al., 2001; Kaste et al., 2003; Klaminder et al., 2005).

Over the past several decades, workers have also developed time-series data for Pb in precipitation, vegetation, organic horizons, mineral soils, and surface waters. Since atmospherically delivered Pb often comprises a significant fraction of the "labile" Pb (i.e., Pb not associated with primary minerals), these data have been useful for developing transport and residence time models of Pb in different terrestrial reservoirs (Friedland et al., 1992; Miller and Friedland, 1994; Johnson et al., 1995b; Wang and Benoit, 1997). Overall, a significant amount of research has been published on the distribution, speciation, and behavior of anthropogenic Pb in the terrestrial environment since 1986. However, certain specific details on the behavior of Pb in the terrestrial environment and its potential effects on soil microorganisms remain elusive.

8.1.3.2 Speciation of Atmospherically-Delivered Lead in Terrestrial Ecosystems

8.1.3.2.1 Lead in the Solid Phases

Selective chemical extractions have been employed extensively over the past 20 years for quantifying amounts of a particular metal phase (e.g., PbS, Pb-humate, Pb-Fe/Mn-oxide) present in soil or sediment rather than total metal concentration. Sometimes selective chemical extractions are applied *sequentially* to a particular sample. For example, the *exchangeable* metal fraction is removed from the soil using a weak acid or salt solution (e.g., $BaCl_2$), followed sequentially by an extraction targeting organic matter (e.g., H_2O_2 or NaOCl), an extraction targeting secondary iron oxides (e.g., $NH_2OH{\cdot}HCl$), and finally, a reagent cocktail targeting primary minerals (e.g., HNO_3-HCl-HF). Tessier et al. (1979) developed this technique. More recently, this technique has been modified and developed specifically for different metals and different types of materials (Keon et al., 2001). Alternatively, batch-style selective chemical extractions have been used on soils and sediments to avoid the problems associated with nonselective reagents (Johnson and Petras, 1998). Selective extractions can be a relatively rapid, simple, and inexpensive means for determining metal phases in soils and sediments, and the generated data can be linked to potential mobility and bioavailability of the metal (Tessier and Campbell, 1987). However, some problems persist with the selective extraction technique. First, extractions are rarely specific to a single phase. For example, while H_2O_2 is often used to remove metals bound to organic matter in soils, others have demonstrated that this reagent destroys clay minerals and sulfides (Ryan et al., 2002). Peroxide solutions may also be inefficient in removing metals bound to humic acids, and in fact could potentially result in the precipitation of metal-humate substances. In addition to the nonselectivity of reagents, significant metal redistribution has been documented to occur during sequential chemical extractions (Ho and Evans, 2000), and many reagents may not completely extract targeted phases. While chemical extractions provide some useful information on metal phases in soil or sediment, the results should be treated as "operationally defined," e.g., "H_2O_2-liberated Pb" rather than "organic Pb."

Lead forms strong coordination complexes with oxygen on mineral surfaces and organic matter functional groups (Abd-Elfattah and Wada, 1981), because of its high electronegativity and hydrolysis constant. Therefore, Pb is generally not readily exchangeable, i.e., the amount of Pb removed from soils by dilute acid or salts is usually less that 10% (Karamanos et al., 1976;

Sposito et al., 1982; Miller and McFee, 1983; Johnson and Petras, 1998; Bacon and Hewitt, 2005). Lead is typically adsorbed to organic and inorganic soil particles strongly via inner-sphere adsorption (Xia et al., 1997; Bargar et al., 1997a,b, 1998). Kaste et al. (2005) found that a single extract of 0.02 M HCl removed 15% or less Pb in organic horizons from a montane forest in New Hampshire. The fact that relatively concentrated acids, reducing agents, oxidizing agents, or chelating agents are required to liberate the majority of Pb from soils is used as one line of evidence that Pb migration and uptake by plants in soils is expected to be low.

Lead that is "organically bound" in soils is typically quantified by extractions that dissolve/disperse or destroy organic matter. The former approach often employs an alkaline solution (NaOH), which deprotonates organic matter functional groups, or a phosphate solution, which chelates structural cations. Extractions used to destroy organic matter often rely on H_2O_2 or NaOCl. Both organic and mineral horizons typically have significant Pb in this soil phase. Miller and McFee (1983) used $Na_4P_2O_7$ to extract organically bound Pb from the upper 2.5 cm of soils sampled from northwestern Indiana. They found that organically bound Pb accounted for between 25 and 50% of the total Pb present in the sampled topsoils. Jersak et al. (1997), Johnson and Petras (1998), and Kaste et al. (2005) selectively extracted Pb from spodosols from the northeastern United States. Using acidified H_2O_2, Jersak et al. (1997) found that very little (<10 %) of the Pb in mineral soils (E, B, C) sampled from New York and Vermont was organic. Johnson and Petras (1998) used $K_4P_2O_7$ to quantify organically bound Pb in the Oa horizon and in mineral soils from the Hubbard Brook Experimental Forest in New Hampshire. They reported that 60% of the total Pb in the Oa horizon was organic and that between 8 and 17% of the total Pb in the mineral soil was organic. However, in the E, Bh, and Bs1 horizons, organically bound Pb dominated the total "labile" (non-mineral lattice) Pb. Kaste et al. (2005) used selective chemical extractions on organic horizons from montane forests in Vermont and New Hampshire. They found that repeated extractions with $Na_4P_2O_7$ removed between 60 and 100% of the Pb from their samples. Caution should be used when interpreting the results of pyrophosphate extractions. Although they are often used to quantify organically-bound metals, this reagent can both disperse and dissolve Fe phases (Jeanroy and Guillet, 1981; Shuman, 1982). Acidified H_2O_2 has also been reported to destroy and release elements associated with secondary soil minerals (Papp et al., 1991; Ryan et al., 2002).

Aside from organic forms, Pb is often found to be associated with secondary oxide minerals in soils. Pb can be partitioned with secondary oxides by a variety of mechanisms, including (1) simple ion exchange, (2) inner-sphere or outer-sphere adsorption, and (3) co-precipitation and/or occlusion (Bargar et al., 1997a,b, 1998, 1999). As discussed above, very little Pb is removed from soil via dilute acid or salt solutions, so adsorption and co-precipitation are likely the dominant Pb interaction with secondary mineral phases. Reagents used to quantify this phase are often solutions of EDTA, oxalate, or hydroxylamine hydrochloride (HH). Miller and McFee (1983) used an EDTA solution followed by an HH solution to quantify Pb occluded by Fe and Mn minerals, respectively, in their surface-soil samples from Indiana. They reported that approximately 30% of the total soil Pb was occluded in Fe minerals, and 5 to 15% was occluded in Mn phases. In soils from the northeastern United States, Jersak et al. (1997) used various strengths of HH solutions and concluded that negligible Pb was associated with Mn-oxides and that 1 to 30% of the Pb was associated with Fe phases in the mineral soils in their study. Johnson and Petras (1998) reported that no Pb was removed from the Oa horizon at the HBEF by oxalate, but that 5 to 15% of the total Pb in mineral soils was removed by this extraction, presumably because it was bound to amorphous oxide minerals. Kaste et al. (2005), however, reported that HH removed 30 to 40% of the Pb from organic horizons in their study. They concluded that Fe phases were important in scavenging Pb, even in soil horizons dominated by organic matter.

Synchrotron radiation (X-rays) allows researchers to probe the electron configuration of metals in untreated soil and sediment samples. This type of analysis has been extremely valuable for determining the coordination environment of Pb in a variety of soils in sediments. Since different elements have different electron binding energies (E_b), X-rays can be focused in an energy window specific to a metal of interest. In experiments involving XAS, X-ray energy is increased until a rapid increase in the amount of absorption occurs; this absorption edge represents E_b. The precise energy required to dislodge a core electron from a metal (i.e., E_b) will be a function of the oxidation state and covalency of the metal. X-ray absorption studies that focus on the location of the absorption edge are referred to as XANES (X-ray absorption near edge structure). In the energy region immediately after the absorption edge, X-ray absorption increases and decreases with a periodicity that represents the wave functions of the ejected electrons and the constructive and destructive interference with the wave functions of the nearby

atoms. X-ray absorption studies used to investigate the periodicity of the absorption after E_b are referred to as EXAFS (extended X-ray absorption fine structure). Since the electron configuration of a Pb atom will be directly governed by its speciation (e.g., Pb bound to organics, Pb adsorbed to oxide surfaces, PbS, etc.) X-ray absorption studies provide a powerful in situ technique for determining speciation without some of the problems associated with chemical extractions (Bargar et al., 1997a,b, 1998).

Manceau et al. (1996) used EXAFS to study soil contaminated by gasoline-derived Pb in France and found that the Pb was divalent and complexed to salicylate and catechol-type functional groups of humic substances. He concluded that the alkyl-tetravalent Pb compounds that were added to gasoline were relatively unstable and will not dominate the speciation of Pb fallout from the combustion of leaded gasoline. The binding mechanism of Pb to organics is primarily inner-sphere adsorption (Xia et al., 1997). DeVolder et al. (2003) used EXAFS to demonstrate that Pb phases were shifting to the relatively insoluble PbS when contaminated wetland soils were treated with sulfate. More recent XAS studies have demonstrated the importance of biomineralization of Pb in soils by bacteria and nematodes (Xia et al., 1997; Templeton et al., 2003a,b; Jackson et al., 2005). Templeton et al. (2003a,b) demonstrated that biogenic precipitation of pyromorphite was the dominant source of Pb uptake by *Burkholderia cepacia* biofilms below pH 4.5. Above pH 4.5, adsorption complexes began to form in addition to Pb mineral precipitation.

In addition to XAS studies of Pb in environmental samples, numerous experimental-based XAS studies have documented in detail the coordination environment of Pb adsorbed to Fe-oxides, Mn-oxides, Al-oxides, and clay minerals (Manceau et al., 1996, 2000a,b, 2002; Bargar et al., 1997a,b, 1998, 1999; Strawn and Sparks, 1999; Trivedi et al., 2003). Bargar et al. (1997a) showed that Pb can adsorb to FeO_6 octahedra on three different types of sites: on corners, edges, or faces. Ostergren et al. (2000a,b) showed that the presence of dissolved carbonate and sulfate increased Pb adsorbtion on goethite. The relative fraction of corner-sharing complexes can be greatly increased by the presence of these ligands, as bridging complexes between the metal and the corners are formed (Ostergren et al., 2000a,b).

Recently, Jackson et al. (2005) used microfocused synchrotron-based X-ray fluorescence (μSXRF) to detail the distribution of Pb and Cu in the nematode *Caenorhabditis elegans.* They found that, while Cu was evenly distributed throughout the bodies of exposed *Caenorhabditis*

elegans, Pb was concentrated in the anterior pharynx region. Microfocused X-ray diffraction indicated that the highly concentrated Pb region in the pharynx was actually comprised of the crystalline Pb mineral, pyromorphite. The authors concluded that *C. elegans* precipitated pyromorphite in the pharynx as a defense mechanism to prevent spreading the toxic metal to the rest of the organism's body. They further suggested that, because of the high turnover rate of nematodes, biomineralization could play an important role in the speciation of Pb in certain soils.

8.1.3.2.2 Lead Solid-Solution Partitioning

The concentration of Pb species dissolved in soil solution is probably controlled by some combination of (a) Pb-mineral solubility equilibria, (b) adsorption reactions of dissolved Pb phases on inorganic surfaces (e.g., crystalline or amorphous oxides of Al, Fe, Si, Mn, etc.; clay minerals), and (c) adsorption reactions of dissolved Pb phases on soil organic matter. Dissolved Pb phases in soil solution can be some combination of Pb^{2+} and its hydrolysis species, Pb bound to dissolved organic matter, and Pb complexes with inorganic ligands such as Cl^- and SO_4^{2-}. Alkaline soils typically have solutions supersaturated with respect to $PbCO_3$, $Pb_3(CO_3)_2(OH)_2$, $Pb(OH)_2$, $Pb_3(PO_4)_2$, $Pb_5(PO_4)_3(OH)$, and $Pb_4O(PO_4)_2$ (Badawy et al., 2002). Pb-phosphate minerals in particular are very insoluble, and thermodynamic data predict that these phases will control dissolved Pb in soil solution under a variety of conditions (Nriagu, 1974; Ruby et al., 1994). However, certain chelating agents, such as dissolved organic matter, can prevent the precipitation of Pb minerals (Lang and Kaupenjohann, 2003).

Soil solution dissolved organic matter content and pH typically have very strong positive and negative correlations, respectively, with the concentration of dissolved Pb species (Sauvé et al., 1998, 2000b, 2003; Weng et al., 2002; Badawy et al., 2002; Tipping et al., 2003). In the case of adsorption phenomena, the partitioning of Pb^{2+} to the solid phase is also controlled by total metal loading, i.e., high Pb loadings will result in a lower fraction being partitioned to the solid phase. Sauvé et al. (1997, 1998) demonstrated that only a fraction of the total Pb in solution was actually Pb^{2+} in soils treated with leaf compost. The fraction of Pb^{2+} to total dissolved Pb ranged from <1 to 60%, depending on pH and the availability of Pb-binding ligands. Nolan et al. (2003) used Donnan dialysis to show that 2.9 to 48.8% of the dissolved Pb was Pb^{2+} in pore waters of agricultural and contaminated soils from Australia and the United

States. In acidic soils, Al species can compete for sites on natural organic matter and inhibit Pb binding to surfaces (Gustafsson et al., 2003).

Differential pulse anodic stripping voltammetry (DPASV) is a technique that is useful for identifying relatively low concentrations of Pb^{2+} and has found many applications in adsorption and partitioning experiments. This technique has been particularly useful for quantifying the K_d, or partitioning ratio of Pb in the solid-to-liquid phase (K_d = [total solid-phase metal in mg kg^{-1}] / [dissolved metal in mg L^{-1}]). While the exact K_d value is a function of pH, organic matter content, substrate type, total metal burden, and concentrations of competing ligands, such studies typically show that Pb has very strong solid-phase partitioning. Partitioning ratios determined by DPASV generally range from 10^3 to 10^6 in soils in the typical pH range (Sauvé et al., 2000a). Aualiitia and Pickering (1987) used thin film ASV to compare the relative affinity of Pb for different inorganic particulates. They reported that Mn(IV) oxides completely adsorbed the Pb, regardless of pH in the range of 3 to 9, and had the highest affinity for Pb in their study. The adsorption of Pb to pedogenic Fe-oxides, Al-hydroxides, clay minerals, and Fe ores was reported to be pH-dependent. Sauvé et al. (1998) used DPASV to study the effects of organic matter and pH on Pb adsorption in an orchard soil. They demonstrated that Pb complexation to dissolved organic matter (DOM) increased Pb solubility, and that 30 to 50% of the dissolved Pb was bound to DOM at pH 3 to 4, while >80% of the dissolved Pb was bound to DOM at neutral pH. They concluded that in most soils, Pb in solution would not be found as Pb^{2+} but as bound to DOM. Sauvé et al. (2000b) compared the relative affinity of Pb^{2+} for synthetic ferrihydrite, leaf compost, and secondary oxide minerals collected from soils. They reported that the inorganic mineral phases were more efficient at lowering the amount of Pb^{2+} that was available in solution than the leaf compost. Glover et al. (2002) used DPSAV in studying the effects of time and organic acids on Pb adsorption to goethite. They found that Pb adsorption to geothite was very rapid, and remained unchanged after a period of about 4 h. Lead desorption was found to be much slower, however, and adsorption was not reversible on a time scale of 8 h. The presence of salicylate appeared to increase the amount of Pb that desorbed from goethite more so than oxalate.

8.1.3.3 Tracing the Fate of Atmospherically Delivered Lead in Terrestrial Ecosystems

Radiogenic Pb isotopes offer a powerful tool for separating anthropogenic Pb from natural Pb derived from mineral weathering (Erel and Patterson, 1994; Erel et al., 1997). This is particularly useful for studying mineral soil, where geogenic Pb often dominates. The three radiogenic stable Pb isotopes (^{206}Pb, ^{207}Pb, and ^{208}Pb) have a heterogeneous distribution in the earth's crust primarily because of the differences in the half-lives of their respective parents (^{238}U, $T_{1/2} = 4.7 \times 10^9$ year; ^{235}U, $T_{1/2} = 0.7 \times 10^9$ year; ^{232}Th, $T_{1/2} = 14 \times 10^9$ year). The result is that the ore bodies from which anthropogenic Pb are typically derived are usually enriched in ^{207}Pb relative to ^{206}Pb and ^{208}Pb when compared with Pb found in granitic rocks. Graney et al. (1995) analyzed a dated core from Lake Erie, and found that the ^{206}Pb:^{207}Pb value in sediment deposited in the late 1700s was 1.224, but in 20th-century sediment, the ratio ranged from 1.223 to 1.197. This shift in the Pb isotopic composition represents the introduction of a significant amount of anthropogenic Pb into the environment. Bindler et al. (1999) and Emmanuel and Erel (2002) analyzed the isotopic composition of Pb in soil profiles in Sweden and the Czech Republic, respectively, and determined that mineral soils immediately below the organic horizon had a mixture of both anthropogenic and geogenic Pb.

Erel and Patterson (1994) used radiogenic Pb isotopes to trace the movement of industrial Pb from topsoils to groundwaters to streams in a remote mountainous region of Yosemite National Park in California. They calculated that total 20th-century industrial Pb input to their study site was approximately 0.4 g Pb m^{-2}. Lead concentrations in organic material were highest in the upper soil horizons, and decreased with depth. During snowmelt, Pb in the snowpack was mixed with the anthropogenic and geogenic Pb already in the topsoil, and spring melts contained a mixture of anthropogenic and geogenic particulate Pb. During base flows, however, 80% of the Pb export from groundwater and streams was from natural granite weathering (Erel and Patterson, 1994).

Uranium-238 series ^{210}Pb also provides a tool for tracing atmospherically delivered Pb in soils. After ^{222}Rn ($T_{1/2}$ = 3.8 days) is produced from the decay of ^{226}Ra ($T_{1/2}$ = 1600 years), some fraction of the ^{222}Rn escapes from rocks and soils to the atmosphere. It then decays relatively rapidly to ^{210}Pb ($T_{1/2}$ = 22.3 years), which has a tropospheric residence time of a few weeks (Koch et al., 1996). Fallout ^{210}Pb is deposited onto forests via wet and dry deposition, similar to anthropogenic Pb deposition in forests, and is thus useful as a tracer for non-native Pb in soils.

Lead-210 is convenient to use for calculating the residence time of Pb in soil layers, because its atmospheric and soil fluxes can be assumed to be in steady state at undisturbed sites (Dörr and Münnich, 1989; Dörr, 1995; Kaste et al., 2003). Atmospheric ^{210}Pb ($^{210}Pb_{ex}$ hereafter, ^{210}Pb in "excess" of that supported by ^{222}Rn in the soil) must be calculated by subtracting the amount of ^{210}Pb formed in soils by the in situ decay of ^{222}Rn from the total ^{210}Pb (Moore and Poet, 1976; Nozaki et al., 1978).

Benninger et al. (1975) measured fallout ^{210}Pb in soils and streamwater at Hubbard Brook and at an undisturbed forest in Pennsylvania. They estimated atmospheric ^{210}Pb export in streamwaters to be <0.02% of the standing ^{210}Pb crop in the organic horizons. They used a simple steady-state model to calculate the residence time of Pb in the organic horizons to be 5,000 years. This overestimate of the Pb residence time in the organic horizons was likely a result of the low resolution of their sampling. Since they only sampled the upper 6 cm of soil and the drainage waters, they did not accurately evaluate the distribution of ^{210}Pb in the soil column in between. Dörr and Münnich (1989, 1991) used ^{210}Pb profiles in soils of southern Germany to evaluate the behavior of atmospherically delivered Pb. They calculated the vertical velocity of Pb by dividing the relaxation depth (i.e., the depth at which ^{210}Pb activity decreases to $1/e$, or approximately 37% of its surface value) by the ^{210}Pb mean life of 32 years. They reported downward transit velocities of atmospherically deposited Pb at 0.89 ± 0.33 mm year^{-1}. The downward transport of atmospheric Pb was not affected by pH or soil type. However, since Pb velocities in the soil profile where identical to carbon velocities calculated with ^{14}C, they concluded that Pb movement in forest soils is probably controlled by carbon transport. Kaste et al. (2003) used ^{210}Pb to model the response time of atmospherically delivered Pb in the O horizon at Camel's Hump Mountain in Vermont. They concluded that the forest floor response time was between 60 and 150 years, depending on vegetation zone and elevation. Using ^{206}Pb:^{207}Pb, they also demonstrated that some gasoline-derived Pb migrated out of the O horizon and into the mineral soil in the deciduous vegetation zone on the mountain, while all of the atmospheric Pb was retained in the upper 20 cm of the soil profile.

Researchers assessing the fate of atmospheric Pb in soils have also relied on repeated sampling of soils and vegetation for total Pb. This technique works best when anthropogenic Pb accounts for the vast majority of total Pb in a particular reservoir. Johnson et al. (1995b), Yanai et al. (2004), and Friedland et al. (1992) used O horizon (forest floor) time-series data to evaluate

the movement of gasoline-derived Pb in the soil profile. These studies have concluded that the distribution of Pb in the upper soil horizons has changed over the past few decades. Yanai et al. (2004) documented a decline in Pb from the Oie horizon between the late 1970s to the early 1990s in remote forest soils in New Hampshire. Johnson et al. (1995b) and Friedland et al. (1992) demonstrated that some fraction of Pb had moved from the O horizon to the mineral soil during the 1980s at Hubbard Brook and at selected remote sites in the northeastern United States, respectively. Evans et al. (2005) demonstrated that Pb concentrations in the litter layer (fresh litter + Oi horizon) sampled in a transect from Vermont to Quebec decreased significantly between 1979 and 1996, reflecting a decrease in Pb deposition to forests and upper soil horizons during that time period. Miller et al. (1993) and Wang and Benoit (1997) used forest floor time-series data to model the response time (*e* folding time, the time it takes a reservoir to decrease to the 1/*e*, (ca. 37%) of its original amount) of Pb in the forest floor. Miller et al. (1993) calculated O horizon response times of 17 years for the northern hardwood forest and 77 years in the spruce-fir zone on Camel's Hump Mountain in Vermont. Wang and Benoit (1997) determined that the O horizon would reach steady state with respect to Pb (1.3 $\mu g\ g^{-1}$ Pb) by 2100. Both suggested that the movement of organic particulates dominated Pb transport in the soil profile.

8.1.3.4 Inputs/Outputs of Atmospherically Delivered Lead in Terrestrial Ecosystems

The concentration of Pb in rainfall in the northeastern United States is on the order of 500 pg g^{-1} (Wang et al., 1995). Assuming a precipitation rate of 1 m $year^{-1}$, then loadings to terrestrial ecosystems via wet deposition are currently on the order of 0.5 mg m^{-2} $year^{-1}$. Since dry deposition may account for anywhere between 10 and 40% of total Pb deposition (Galloway et al., 1982), total loadings to ecosystems are approximately 1 mg m^{-2} $year^{-1}$. This is a relatively small annual flux of Pb if compared to the reservoir of approximately 0.5 to 4 g m^{-2} of gasoline-derived Pb that is already in surface soils over much of the United States (Friedland et al., 1992; Miller and Friedland, 1994; Erel and Patterson, 1994; Marsh and Siccama, 1997; Yanai et al., 2004; Johnson et al., 2004; Evans et al., 2005). While vegetation can play an important role in sequestering Pb from rain and dry deposition (Russell et al., 1981), direct uptake of Pb from soils by plants appears to be low (Klaminder et al., 2005). High elevation areas, particularly those near the base level of clouds often have higher burdens of atmospheric contaminants (Siccama, 1974). A Pb deposition model by Miller and Friedland (1994) predicted 2.2 and 3.5 g Pb m^{-2}

deposition for the 20th century in the deciduous zone (600 m) and the coniferous zone (1000 m), respectively. More recently, Kaste et al. (2003) used radiogenic isotope measurements on the same mountain to confirm higher loadings at higher elevation. They measured 1.3 and 3.4 g gasoline-derived Pb m^{-2} in the deciduous zone and coniferous zones, respectively. Higher atmospheric Pb loadings to higher elevations are attributed to (1) the higher leaf area of coniferous species, which are generally more prevalent at high elevation; (2) higher rainfall at higher elevation; and (3) increased cloudwater impaction at high elevation (Miller et al., 1993).

Although inputs of Pb to ecosystems are currently low, Pb export from watersheds via groundwater and streams is substantially lower. Therefore, even at current input levels, watersheds are accumulating industrial Pb. Seeps and streams at the HBEF have Pb concentrations on the order of 10 to 30 pg Pb g^{-1} (Wang et al., 1995). At a remote valley in the Sierra Nevada, Pb concentrations in streamwaters were on the order of 15 pg Pb g^{-1} (Erel and Patterson, 1994). Losses of Pb from soil horizons are assumed to be via particulates (Dörr and Münnich, 1989; Wang and Benoit, 1996, 1997). Tyler (1981) noted that Pb losses from an a horizon in Sweden were influenced by season; with highest Pb fluxes being observed during warm, wet months. He suggested that DOC production and Pb movement were tightly linked.

Surface soils across the United States are enriched in Pb relative to levels expected from solely natural geogenic inputs (Friedland et al., 1984; Francek, 1992; Erel and Patterson, 1994; Marsh and Siccama, 1997; Yanai et al., 2004; Murray et al., 2004). While some of this contaminant Pb is attributed to paint, salvage yards, and the use of Pb-arsenate as a pesticide in localized areas (Francek, 1997), Pb contamination of surface soils is essentially ubiquitous because of atmospheric pollution associated with the metal production industry and the combustion of fossil fuels. Surface soils in Michigan, for example, typically range from 8 to several hundred ppm Pb (Francek, 1992; Murray et al., 2004). Soils collected and analyzed beneath 50 cm in Michigan, however, range only from 4 to 60 ppm Pb (Murray et al., 2004). In remote surface soils from the Sierra Nevada Mountains, litter and upper soil horizons are 20 to 40 ppm Pb, and approximately 75% of this Pb has been attributed to atmospheric deposition during the 20th century (Erel and Patterson, 1994). Repeated sampling of the forest floor (O horizon) in the northeastern United States demonstrates that the organic layer has retained much of the Pb load deposited on ecosytems during the 20th century. Total Pb deposition during the 20th century has been estimated at 1 to 3 g Pb m^{-2}, depending on elevation and proximity to

urban areas (Miller and Friedland, 1994; Johnson et al., 1995b). Forest floors sampled during the 1980s and 1990s, and in early 2000 had between 0.7 and 2 g Pb m^{-2} (Friedland et al., 1992; Miller and Friedland, 1994; Johnson et al., 1995b; Kaste et al., 2003; Yanai et al., 2004; Evans et al., 2005). The pool of Pb in above- and below-ground biomass at the HBEF is on the order of 0.13 g Pb m^{-2} (Johnson et al., 1995b).

The amount of Pb that has leached into mineral soil appears to be on the order of 20 to 50% of the total anthropogenic Pb deposition. Kaste et al. (2003) and Miller and Friedland (1994) demonstrated that Pb loss from the forest floor at Camel's Hump Mountain in Vermont depended on elevation. While the mineral soil in the deciduous forest had between 0.4 and 0.5 g Pb m^{-2} (out of 1 to 2 g Pb m^{-2} in the total soil profile), at higher elevations the thicker coniferous forest floor retained more than 90% of the total Pb deposition (Kaste et al., 2003). Johnson et al. (1995b) determined that the forest floor at HBEF in the mid-1980s had about 0.75 g Pb m^{-2}. Compared to their estimated 20th-century atmospheric Pb deposition of 0.9 g Pb m^{-2}, the forest floor has retained 83% of the atmospheric Pb loadings (Johnson et al., 1995b). Johnson et al. (2004) noted that gasoline-derived Pb was a significant component of the labile Pb at the HBEF. They calculated that Pb fluxes to the HBEF by atmospheric pollution were essentially equivalent to the Pb released by mineral weathering over the past 12,000 years. Marsh and Siccama (1997) used the relatively homogenous mineral soils underneath formerly plowed land in New Hampshire, Connecticut, and Rhode Island to assess the depth-distribution of atmospheric Pb. They reported that 65% of the atmospheric Pb deposited during the 20th century is in the mineral soil and 35% is in the forest floor. At their remote study site in the Sierra Nevada Mountains, Erel and Patterson (1994) reported that most of the anthropogenic Pb was associated with the humus fraction of the litter layer and soils sampled in the upper few cm.

Atmospherically delivered Pb is probably present in ecosystems in a variety of different biogeochemical phases. A combination of Pb adsorbtion processes and the precipitation of Pb minerals will typically keep dissolved Pb species low in soil solution, surface waters, and streams (Sauvé et al., 2000b; Jackson et al., 2005). While experimental and theoretical evidence suggest that the precipitation of inorganic Pb phases and the adsorption of Pb on inorganic phases can control the biogeochemistry of contaminant Pb (Nriagu, 1974; Ruby et al., 1994; Jackson et al., 2005), the influence of organic matter on the biogeochemistry of Pb in terrestrial ecosystems cannot be ignored in many systems. Organic matter can bind to Pb, preventing Pb

migration and the precipitation of inorganic phases (Manceau et al., 1996; Xia et al., 1997; Lang and Kaupenjohann, 2003). As the abundance of organic matter declines in soil, Pb adsorption to inorganic soil minerals and the direct precipitation of Pb phases may dominate the biogeochemistry of Pb in terrestrial ecosystems (Ostergren et al., 2000a,b; Sauvé et al., 2000b).

Conclusions

Advances in technology since the 1986 Pb AQCD have allowed for a quantitative determination of the mobility, distribution, uptake, and fluxes of atmospherically delivered Pb in ecosystems. Among other things, these studies have shown that industrial Pb represents a significant fraction of total labile Pb in watersheds. Selective chemical extractions and synchrotron-based X-ray studies have shown that industrial Pb can be strongly sequestered by organic matter and by secondary minerals such as clays and oxides of Al, Fe, and Mn. Some of these studies have provided compelling evidence that the biomineralization of Pb phosphates by soil organisms can play an important role in the biogeochemistry of Pb. Surface soils sampled relatively recently demonstrate that the upper soil horizons (O + A horizons) are retaining most of the industrial Pb burden introduced to the systems during the 20th century. The migration and biological uptake of Pb in ecosystems is relatively low. The different biogeochemical behaviors of Pb reported by various studies may be a result of the many different analytical techniques employed, or they may be a result of natural variability in the behavior of Pb in different systems.

8.1.4 Species Response/Mode of Action

8.1.4.1 Introduction

The 1986 Pb AQCD, Volume II (U.S. Environmental Protection Agency, 1986a) reviewed the literature on the uptake of Pb into plants, soil organisms, birds, and mammals. This chapter expands upon the major conclusions from the EPA (U.S. Environmental Protection Agency, 1986a) related to those organisms. It summarizes the recent (since 1986) critical research conducted on Pb uptake into terrestrial organisms (Section 8.1.4.2), mechanisms of resistance to Pb toxicity (Section 8.1.4.3), the physiological effects of Pb (Section 8.1.4.4), and, the factors that modify organism response to Pb (Section 8.1.4.5). A summary is presented

in Section 8.1.4.6. All concentrations are expressed as mg Pb/kg dw (dry weight) unless otherwise indicated.

Areas of research that are not addressed include those that used irrelevant exposure conditions relative to airborne emissions of Pb (e.g., Pb shot, Pb paint, injection studies, studies conducted on mine tailings or using hyperaccumulator plants for phytoremediation, and studies conducted with hydroponic solutions) except when these studies provided critical information for understanding physiologic effects.

8.1.4.2 Lead Uptake

Since the 1986 Pb AQCD, there have been several studies that evaluated the uptake of Pb into plants and invertebrates. The mechanisms associated with Pb uptake and translocation are described in this section. The methods used by the EPA (U.S. Environmental Protection Agency, 2005b) to estimate Pb uptake into plants, earthworms, and small mammals as part of Ecological Soil Screening Level (Eco-SSL) development are also presented.

The accumulation of Pb into the various tissues of consumers (birds and mammals) is discussed only when it was described relative to either environmental concentrations or organismal effects. Numerous other monitoring studies measuring only the Pb concentrations in various tissues of birds and mammals were not included in this chapter; their data cannot be used to develop an air standard without information on environmental concentrations or organismal effects.

Lead Uptake into Plants

Plants take up Pb via their foliage and through their root systems (U.S. Environmental Protection Agency, 1986a; Påhlsson, 1989). Surface deposition of Pb onto plants may represent a significant contribution to the total Pb in and on the plant, as has been observed for plants near smelters and along roadsides (U.S. Environmental Protection Agency, 1986a). The importance of atmospheric deposition on above-ground plant Pb uptake is well-documented (Dalenberg and Van Driel, 1990; Jones and Johnston, 1991; Angelova et al., 2004). Data examined from experimental grassland plots in southeast England demonstrated that atmospheric Pb is a greater contributor than soil-derived Pb in crop plants and grasses (Jones and Johnston, 1991). A study by Dalenberg and Van Driel (1990) showed that 75 to 95% of the Pb found in field-grown test

plants (i.e., the leafy material of grass, spinach, and carrot; wheat grain; and straw) was from atmospheric deposition. Angelova et al. (2004) found that tobacco grown in an industrial area accumulated significant amounts of Pb from the atmosphere, although uptake from soil was also observed. The concentration of Pb in tobacco seeds was linearly related to the concentration of Pb in the exchangeable and carbonate-bound fractions of soil, as measured using sequential extraction (Angelova et al., 2004). Lead in soil is more significant when considering uptake into root vegetables (e.g., carrot, potato), since, as was noted in the 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a), most Pb remains in the roots of plants.

There are two possible mechanisms (symplastic or apoplastic) by which Pb may enter the root of a plant. The symplastic route is through the cell membranes of root hairs; this is the mechanism of uptake for water and nutrients. The apoplastic route is an extracellular route between epidermal cells into the intercellular spaces of the root cortex. Previously, Pb was thought to enter the plant via the symplastic route, probably by transport mechanisms similar to those involved in the uptake of calcium or other divalent cations (i.e., transpirational mass flow, diffusion, or active transport). However, it also had been speculated that Pb may enter the plant via the apoplastic route (U.S. Environmental Protection Agency, 1986a). Sieghardt (1990) determined that the mechanism of Pb uptake was via the symplastic route only and that the apoplastic pathway of transport was stopped in the primary roots by the endodermis. He studied the uptake of Pb into two plants, *Minuartia verna* (moss sandwort) and *Silene vulgaris* (bladder campion) that colonize metal-contaminated sites. In the roots of both plants, Pb was found mainly in the root cortex. Active ion uptake was required to transport the Pb into the stele and then into the shoots of the plant (Sieghardt, 1990).

Although some plants translocate more Pb to the shoots than others, most Pb remains in the roots of plants. Two mechanisms have been proposed to account for this relative lack of translocation to the shoots: (1) Pb may be deposited within root cell wall material, or (2) Pb may be sequestered within root cell organelles (U.S. Environmental Protection Agency, 1986a). Påhlsson (1989) noted that plants can accumulate large quantities of Pb from the soil but that translocation to shoots and leaves is limited by the binding of Pb ions at root surfaces and cell walls. In a study by Wierzbicka (1999), 21 different plant species were exposed to Pb^{2+} in the form of Pb-chloride. The plant species included cucumber (*Cucumis sativus*), soy bean (*Soja hispida*), bean (*Phaseolus vulgaris*), rapeseed (*Brassica napus*), rye (*Secale cereale*), barley

(*Hordeum vulgare*), wheat (*Triticum vulgare*), radish (*Raphanus sativus*), pea (*Pisum sativum*), maize (*Zea mays*), onion (*Allium cepa*), lupine (*Lupinus luteus*), bladder campion (*Silene vulgaris*), Buckler mustard (*Biscutella laevigata*), and rough hawkbit (*Leontodon hispidus*). Although, the amount of Pb taken up by the plant varied with species, over 90% of absorbed Pb was retained in the roots. Only a small amount of Pb was translocated (~2 to 4%) to the shoots of the plants. Lead in roots was present in the deeper layers of root tissues (in particular, the root cortex) and not only on the root surface. There was no correlation between Pb tolerance (measured as root mass increase expressed as a percentage of controls) and either root or shoot tissue concentrations (Wierzbicka, 1999). The study by Wierzbicka (1999) was the first to report that plants developing from bulbs, in this case the onion, were more tolerant to Pb than plants developing from seeds. This tolerance was assumed to be related to the large amounts of Pb that were transported from the roots and stored in the bulb of the plant (Wierzbicka, 1999).

Uptake of Pb from soil into plants was modeled as part of Eco-SSL development (U.S. Environmental Protection Agency, 2005b). The relationship derived between Pb in the soil and Pb in a plant was taken from Bechtel Jacobs Company (BJC) (1998) and is as follows:

$$\text{Ln}(Cp) = 0.561 * \text{Ln}(Csoil) - 1.328 \qquad (8\text{-}1)$$

where Cp is the concentration of Pb in the plant (dry weight) and Csoil is the concentration of Pb in the soil. This equation recognizes that the ratio of Pb concentration in plant to Pb concentration in soil is not constant.

Invertebrates

There was no clear evidence suggesting a differential uptake of Pb into different species of earthworm (*Lumbricus terrestris*, *Aporrectodea rosea*, and *A. caliginosa*) collected around a smelter site near Avonmouth, England (Spurgeon and Hopkin, 1996a). This is in contrast to Pižl and Josens (1995) and Terhivuo et al. (1994) who found *Aporrectodea* spp. accumulated more Pb than *Lumbricus*. The authors suggested that these differences could be due to different feeding behaviors, as *Lumbricus* feeds on organic material and *Apporectodea* species are geophagus, ingesting large amounts of soil during feeding. The differences between species also may be related to differing efficiencies in excretory mechanisms (Pižl and Josens, 1995).

However, the interpretation of species difference is complicated by a number of potentially confounding variables, such as soil characteristics (e.g., calcium or other nutrient levels) (Pižl and Josens, 1995).

The bioaccumulation of Pb from contaminated soil was tested using the earthworm *Eisenia fetida*, and the amount of Pb accumulated did not change significantly until the concentration within soil reached 5000 mg/kg (Davies et al., 2003). This coincided with the lowest soil concentrations at which earthworm mortality was observed. The ratio of the concentration of Pb in worms to the concentration in soil decreased from 0.03 at 100 mg/kg to 0.001 at 3000 mg/kg, but then increased quickly to 0.02 at 5000 mg/kg. The authors concluded that earthworms exhibit regulated uptake of Pb at levels of low contamination (<3000 mg/kg) until a critical concentration is reached, at which point this mechanism breaks down, resulting in unregulated accumulation and mortality. This study was conducted using test methods where soil was not allowed to equilibrate following the addition of Pb and prior to the addition of the test organisms. This may have resulted in an increased bioavailability and overestimated Pb toxicity relative to actual environmental conditions (Davies et al., 2003).

Lock and Janssen (2002) and Bongers et al. (2004) found that Pb-nitrate was more toxic than Pb-chloride to survival and reproduction of the springtail *Folsomia candida*. However, percolation (removal of the chloride or nitrate counterion) caused a significant decrease in Pb-nitrate toxicity such that there was no difference in toxicity once the counterion was removed (Bongers et al., 2004). No change in toxicity was observed for Pb-chloride once the chloride was removed from the soil. Bongers et al. (2004) suggested that the nitrate ion was more toxic than the chloride ion to springtails.

Uptake of Pb from soil into earthworms was also modeled as part of Eco-SSL development (U.S. Environmental Protection Agency, 2005b). The relationship derived between Pb in the soil and Pb in an earthworm was taken from Sample et al. (1999) and is as follows:

$$\text{Ln(Cworm)} = 0.807 * \text{Ln(Csoil)} - 0.218 \qquad (8\text{-}2)$$

where Cworm is the concentration of Pb in the earthworm (dry weight) and Csoil is the concentration of Pb in the soil. This equation recognizes that the ratio of Pb concentration in worm to Pb concentration in soil is not constant.

Wildlife

Research has been conducted to determine what Pb concentrations in various organs would be indicative of various levels of effects. For example, Franson (1996) compiled data to determine what residue levels were consistent with three levels of effects in Falconiformes (e.g., falcons, hawks, eagles, kestrels, ospreys), Columbiformes (e.g., doves, pigeons), and Galliformes (e.g., turkey, pheasant, partridge, quail, chickens). The three levels of effect were (1) subclinical, which are physiological effects only, such as the inhibition of δ-aminolevulinic acid dehydratase (ALAD; see Section 8.1.4.4); (2) toxic, a threshold level marking the initiation of clinical signs, such as anemia, lesions in tissues, weight loss, muscular incoordination, green diarrhea, and anorexia; and (3) compatible with death, an approximate threshold value associated with death in field, captive, and/or experimental cases of Pb poisoning. The tissue Pb levels associated with these levels of effects are presented in Table 8-1.4.1.

Table 8-1.4.1. Tissue Lead Levels in Birds Causing Effects (taken from Franson, 1996)

Order	Blood (µg/dL)	Liver (ppm wet wt.)	Kidney (ppm wet wt.)
Falconiformes			
Subclinical	0.2 – 1.5	2 – 4	2 – 5
Toxic	>1	>3	>3
Compatible with death	>5	>5	>5
Columbiformes			
Subclinical	0.2 – 2.5	2 – 6	2 – 20
Toxic	>2	>6	>15
Compatible with death	>10	>20	>40
Galliformes			
Subclinical	0.2 – 3	2 – 6	2 – 20
Toxic	>5	>6	>15
Compatible with death	>10	>15	>50

Tissue residue levels below the subclinical levels in Table 8-1.4.1 should be considered "background" (Franson, 1996). Levels in the subclinical range are indicative of potential injury from which the bird would probably recover if Pb exposure was terminated. Toxic residues

could lead to death. Residues above the compatible-with-death threshold are consistent with Pb-poisoning mortality (Franson, 1996). Additional information on residue levels for Passeriformes (e.g., sparrows, starlings, robins, cowbirds), Charadriiformes (e.g., gulls, terns), Gruiformes (e.g., cranes), Ciconiformes (e.g., egrets), Gaviformes (e.g., loons), and Strigiformes (e.g., owls) is available (Franson, 1996). Scheuhammer (1989) found blood Pb concentrations of between 0.18 and 0.65 µg/mL in mallards corresponded to conditions associated with greater than normal exposure to Pb but that should not be considered Pb poisoning.

Lead concentrations in various tissues of mammals also have been correlated with toxicity (Ma, 1996). The tissues commonly analysed for Pb are blood, liver, and kidney. Typical baseline levels of blood Pb are approximately 4 to 8 µg/dL for small mammals, and 2 to 6 µg/dL for mature cattle. Typical baseline levels of Pb in liver are 1 to 2 mg/kg dw for small mammals. Typical baseline levels of Pb in kidney are 0.2 to 1.5 mg/kg dw for mice and voles, but shrews typically have higher baseline levels of 3 to 19 mg/kg dw. Ma (1996) concluded that Pb levels less than 5 mg/kg dw in liver and 10 mg/kg dw in kidney were not associated with toxicity, but that levels greater than 5 mg/kg dw in liver and greater than 15 mg/kg dw in kidney could be taken as a chemical biomarker of toxic exposure to Pb in mammals. Humphreys (1991) noted that the concentrations of Pb in liver and kidney can be elevated in animals with normal blood Pb concentrations (and without exhibiting clinical signs of Pb toxicity), because Pb persists in these organs longer than in blood.

Uptake of Pb from soil into small mammals was also modeled as part of Eco-SSL development (U.S. Environmental Protection Agency, 2005b). The relationship derived between Pb in the soil and Pb in the whole-body of a small mammal was taken from Sample et al. (1998) and is as follows:

$$Ln(Cmammal) = 0.4422 * Ln(Csoil) + 0.0761 \quad (8\text{-}3)$$

where Cmammal is the concentration of Pb in small mammals (dry weight) and Csoil is the concentration of Pb in the soil. This equation recognizes that the ratio of Pb concentration in small mammals to Pb concentration in soil is not constant.

8.1.4.3 Resistance Mechanisms

Many mechanisms related to heavy metal tolerance in plants and invertebrates have been described, including avoidance (i.e., root redistribution, food rejection), exclusion (i.e., selective uptake and translocation), immobilization at the plant cell wall, and excretion (i.e., foliar leakage, moulting) (Tyler et al., 1989; Patra et al., 2004). The following section reviews the recent literature on the resistance mechanisms of plants and invertebrates through mitigation of Pb (1) toxicity (8.1.4.3.1) or (2) exposure (8.1.4.3.2).

8.1.4.3.1 Detoxification Mechanisms

Lead sequestration in cell walls may be the most important detoxification mechanism in plants. Calcium may play a role in this detoxification by regulating internal Pb concentrations through the formation of Pb-containing precipitates in the cell wall (Antosiewicz, 2005). Yang et al. (2000) screened 229 varieties of rice (*Oryza sativa*) for tolerance or sensitivity to Pb and found that the oxalate content in the root and root exudates was increased in Pb-tolerant varieties. The authors suggested that the oxalate reduced Pb bioavailability, and that this was an important tolerance mechanism (Yang et al., 2000). Sharma et al. (2004) found Pb-sulfur and Pb-sulfate in the leaves, and Pb-sulfur in the roots of *Sesbania drummondii* (Rattlebox Drummond), a Pb hyperaccumulator plant grown in Pb-nitrate solution. They hypothesized that these sulfur ligands were indicative of glutathione and phytochelatins, which play a role in heavy metal homeostasis and detoxification (Sharma et al., 2004).

Sea pinks (*Armeria maritima*) grown on a metal-contaminated site (calamine spoils more than 100 years old) accumulated 6× the concentrations of Pb in brown (dead and withering) leaves than green leaves (Szarek-Lukaszewska et al., 2004). The concentration of Pb in brown leaves was similar to that in roots. This greater accumulation of Pb into older leaves was not observed in plants grown hydroponically in the laboratory. The authors hypothesized that this sequestering of Pb into the oldest leaves was a detoxification mechanism (Szarek-Lukaszewska et al., 2004).

Terrestrial invertebrates also mitigate Pb toxicity. Wilczek et al. (2004) studied two species of spider, the web-building *Agelena labyrinthica* and the active hunter wolf spider *Pardosa lugubris*. The activity of metal detoxifying enzymes (via the glutathione metabolism pathways) was greater in *A. labyrinthica* and in females of both species (Wilczek et al., 2004).

Marinussen et al. (1997) found that earthworms can excrete 60% of accumulated Pb very quickly once exposure to Pb-contaminated soils has ended. However, the remainder of the body burden is not excreted, possibly due to the storage of Pb in waste nodules that are too large to be excreted (Hopkin, 1989). Gintenreiter et al. (1993) found that Lepidoptera larvae (in this case, the gypsy moth *Lymantria dispar*) eliminated Pb, to some extent, in the meconium (the fluid excreted shortly after emergence from the chrysalis).

Lead, in the form of pyromorphite ($Pb_5(PO_4)_3Cl$), was localized in the anterior pharynx region of the nematode *Ceanorhabditis elegans* (Jackson et al., 2005). The authors hypothesized that the nematode may detoxify Pb via its precipitation into pyromorphite, which is relatively insoluble (Jackson et al., 2005).

8.1.4.3.2 Avoidance Response

Studies with soil invertebrates hypothesize that these organisms may avoid soil with high Pb concentrations. For example, Bengtsson et al. (1986) suggested that the lower Pb concentrations in earthworm tissues may be a result of lowered feeding activity of worms at higher Pb concentrations in soil.

8.1.4.4 Physiological Effects of Lead

Several studies have measured decreased blood ALAD activity in birds and mammals exposed to Pb (U.S. Environmental Protection Agency, 1986a). Recent studies on the physiological effects of Pb to consumers have focused on heme synthesis (as measured by ALAD activity and protoporphyrin concentration), lipid peroxidation, and production of fatty acids. Effects on growth are covered in Section 8.1.5.

Biochemically, Pb adversely affects hemoglobin synthesis in birds and mammals. Early indicators of Pb exposure in birds and mammals include decreased blood ALAD concentrations and increased protoporphyrin IX activity. The effects of Pb on blood parameters and the use of these parameters as sensitive biomarkers of exposure has been well documented (Eisler, 1988; U.S. Environmental Protection Agency, 2005a). However, the linkage between these biochemical indicators and ecologically-relevant effects is less well understood. Low-level inhibition of ALAD is not generally considered a toxic response, because this enzyme is thought

to be present in excess concentrations; rather, it may simply indicate that the organism has recently been exposed to Pb (Henny et al., 1991).

Schlick et al. (1983) studied ALAD inhibition in mouse bone marrow and erythrocytes. They estimated that an absorbed dose of between 50 and 100 μg Pb-acetate/kg body weight per day would result in long-term inhibition of ALAD.

Beyer et al. (2000) related blood Pb to sublethal effects in waterfowl along the Coeur d'Alene River near a mining site in Idaho. The sublethal effects measured included, among others, red blood cell ALAD activity and protoporphyrin levels in the blood. As found in other studies, ALAD activity was the most sensitive indicator of Pb exposure, decreasing to 3% of the reference value at a blood Pb concentration of 0.68 mg/kg ww (wet weight). Protoporphyrin concentrations showed a 4.2-fold increase at this same concentration.

Henny et al. (1991) studied osprey along the Coeur d'Alene River. There were no observations of death, behavioral abnormalities, or reduced productivity related to Pb exposure, although inhibition of blood ALAD and increased protoporphyrin concentrations were measured in ospreys. Henny et al. (1991) hypothesized that no impacts to osprey were observed, even though swan mortality was documented in the area because swans feed at a lower trophic level (i.e., Pb does not biomagnify, and thus is found at higher concentrations in lower trophic level organisms).

Hoffman et al. (2000a) also studied the effects of Coeur d'Alene sediment on waterfowl, focusing on mallard ducklings for 6 weeks after hatching. The study revealed that a 90% reduction in ALAD activity and a greater than 3-fold increase in protoporphyrin concentration occurred when blood Pb reached a concentration of 1.41 mg/kg ww as a result of the ducklings being fed a diet composed of 12% sediment (3449 mg/kg Pb). Those ducklings fed a diet composed of 24% sediment were found to have a mean blood Pb concentration of 2.56 mg/kg ww and a greater than 6-fold increase in protoporphyrin concentration. Hoffman et al. (2000b) also studied Canada Geese (*Branta canadensis*) goslings in a similar fashion. The results revealed that, while blood Pb concentrations in goslings were approximately half (0.68 mg/kg ww) of those found in ducklings under the same conditions (12% diet of 3449 mg/kg sediment Pb), goslings showed an increased sensitivity to Pb exposure. Goslings experienced a 90% reduction in ALAD activity and a 4-fold increase in protoporphyrin concentration, similar to conditions found in the ducklings, although blood Pb concentrations were half those found in the

ducklings. More serious effects were seen in the goslings when blood Pb reached 2.52 mg/kg, including decreased growth and mortality.

Redig et al. (1991) reported a hawk LOAEL of 0.82 mg/kg-day for effects on heme biosynthetic pathways. Lead dosages as high as 1.64 to 6.55 mg/kg-day caused neither mortality nor clinical signs of toxicity. A dose of 6.55 mg/kg-day resulted in blood Pb levels of 1.58 µg/ml. There were minimal changes in immune function (Redig et al., 1991).

Repeated oral administration of Pb resulted in biochemical alterations in broiler chickens (Brar et al., 1997a,b). At a dose of 200 mg/kg-day Pb-acetate, there were significant increases in plasma levels of uric acid and creatinine and significant declines in the levels of total proteins, albumin, glucose, and cholesterol. Brar et al. (1997a) suggested that increased uric acid and creatinine levels could be due to an accelerated rate of protein catabolism and/or kidney damage. They also suggested that the decline in plasma proteins and albumin levels may be caused by diarrhea and liver dysfunction due to the Pb exposure. Brar et al. (1997b) also found that significant changes in plasma enzymes may be causing damage to other organs.

Lead can cause an increase in tissue lipid peroxides and changes in glutathione concentrations, which may be related to peroxidative damage of cell membranes (Mateo and Hoffman, 2001). There are species-specific differences in resistance to oxidative stress (lipid peroxidation), which may explain why Canada geese are more sensitive to Pb poisoning than mallards (Mateo and Hoffman, 2001). Lead also caused an increase in the production of the fatty acid arachidonic acid, which has been associated with changes in bone formation and immune response (Mateo et al., 2003a). The effects observed by Mateo et al. (2003a,b) were associated with very high concentrations of Pb in the diet (1840 mg Pb/kg diet), much higher than would be found generally in the environment, and high enough that birds decreased their food intake.

Lead also induces lipid peroxidation in plants. Rice plants exposed to a highly toxic level of Pb (1000 µM in nutrient solution) showed elevated levels of lipid peroxides, increased activity of superoxide dismutase, guaiacol peroxidase, ascorbate peroxidase, and glutatione reductase (Verma and Dubey, 2003). The elevated levels of these enzymes suggest the plants may have an antioxidative defense mechanism against oxidative injury caused by Pb (Verma and Dubey, 2003).

8.1.4.5 Factors that Modify Organism Response

Research has demonstrated that Pb may affect survival, reproduction, growth, metabolism, and development in a wide range of species. These effects may be modified by chemical, biological, and physical factors. The factors that modify responses of organisms to Pb are described in the following sections.

8.1.4.5.1 Genetics

Uptake and toxicity of Pb to plants are influenced strongly by the type of plant. Liu et al. (2003) found that Pb uptake and translocation by rice plants differed by cultivar (a cultivated variety of plant produced by selective breeding) but was not related to genotype. Twenty cultivars were tested from three genotypes. The differences in Pb concentrations among cultivars were smallest when comparing concentrations in the grains at the ripening stage. This study also found that toxicity varied by cultivar; at 800 mg Pb/kg soil, some cultivars were greatly inhibited, some were significantly improved, and others showed no change.

Dearth et al. (2004) compared the response of Fisher 344 (F344) rats and Sprague-Dawley (SD) rats to exposure via gavage to 12 mg Pb/mL as Pb-acetate. Blood Pb levels in the F344 dams were higher than those of the SD dams. Lead delayed the timing of puberty and suppressed hormone levels in F344 offspring. These effects were not observed in the offspring of SD rats, even when the dose was doubled. The authors conclude that F344 rats are more sensitive to Pb (Dearth et al., 2004).

8.1.4.5.2 Biological Factors

Several biological factors may influence Pb uptake and organisms response, including organism age, sex, species, feeding guild, and, for plants, the presence of mycorrhizal fungi. Monogastric animals are more sensitive to Pb than ruminants (Humphreys, 1991).

Younger organisms may be more susceptible to Pb toxicity (Eisler, 1988; Humphreys, 1991). Nestlings are more sensitive to the effects of Pb than older birds, and young altricial birds (species unable to self-regulate body heat at birth, such as songbirds), are considered more sensitive than precocial birds (species that have a high degree of independence at birth, such as quail, ducks, and poultry) (Scheuhammer, 1991).

Sex can also have an effect on the accumulation of Pb by wildlife (Eisler, 1988). Female birds accumulate more Pb than males (Scheuhammer, 1987; Tejedor and Gonzalez, 1992). These and other authors have related this to the increased requirement for calcium in laying females.

Different types of invertebrates accumulate different amounts of Pb from the environment (U.S. Environmental Protection Agency, 1986a). There may be species- and sex-specific differences in accumulation of Pb into invertebrates, specifically arthropods. This has been shown by Wilczek et al. (2004) who studied two species of spider, the web-building *A. labyrinthica* and the active hunter wolf spider *P. lugubris*. The body burdens of Pb in the wolf spider were higher than in the web-building spider, and this may be due to the more effective use of glutathione metabolism pathways in *A. labyrinthica*. Body burdens of females were lower than those of males in both species. This was also observed in spiders by Rabitsch (1995b). Females are thought to be able to detoxify and excrete excess metals more effectively than males (Wilczek et al., 2004). Lead accumulation has been measured in numerous species of arthropods with different feeding strategies. Differences were observed between species (Janssen and Hogervorst, 1993; Rabitsch, 1995a) and depending upon sex (Rabitsch, 1995a), developmental stage (Gintenreiter et al., 1993; Rabitsch, 1995a), and season (Rabitsch, 1995a).

Uptake of Pb may be enhanced by symbiotic associations between plant roots and mycorrhizal fungi. Similar to the mechanism associated with increased uptake of nutrients, mycorrhizal fungi also may cause an increase in the uptake of Pb by increasing the surface area of the roots, the ability of the root to absorb particular ions, and the transfer of ions through the soil (U.S. Environmental Protection Agency, 1986a). There have been contradictory results published in the literature regarding the influence of mycorrhizal organisms on the uptake and toxicity of Pb to plants (see review in Påhlsson, 1989). Lin et al. (2004) found that the bioavailability of Pb increased in the rhizosphere of rice plants, although the availability varied with Pb concentration in soil. Bioavailability was measured as the soluble plus exchangeable Pb fraction from sequential extraction analysis. The authors hypothesized that the enhanced solubility of Pb may be due to a reduced pH in the rhizosphere or, more likely, the greater availability of organic ligands, which further stimulates microbial growth (Lin et al., 2004). Increased bioavailability of Pb in soil may increase the uptake of Pb into plants, although this was not assessed by Lin et al. (2004). However, Dixon (1988) found that red oak

(*Quercus rubra*) seedlings with abundant ectomycorrhizae had lower Pb concentrations in their roots than those seedlings without this fungus, although only at the 100 mg Pb/kg sandy loam soil concentration (no differences were found at lower Pb concentrations). Lead in soil also was found to be toxic to the ectomycorrhizal fungi after 16 weeks of exposure to 50 mg Pb/kg or more (Dixon, 1988). Malcova and Gryndler (2003) showed that maize root exudates from mycorrhizal fungi can ameliorate heavy metal toxicity until a threshold metal concentration was surpassed. This may explain the conflicting results in the past regarding the uptake and toxicity of Pb to plants with mycorrhizal fungi.

The type of food eaten is a major determinant of Pb body burdens in small mammals, with insectivorous animals accumulating more Pb than herbivores or granivores (U.S. Environmental Protection Agency, 1986a). In fact, the main issue identified by the EPA (U.S. Environmental Protection Agency, 1986a) related to invertebrate uptake of Pb was not toxicity to the invertebrates, but accumulation of Pb to levels that may be toxic to their consumers. Several authors suggest that shrews are a good indicator of metal contamination, because they tend to accumulate higher levels of metals than herbivorous small mammals (see data summary in Sample et al. (1998)). Shrews accumulate higher levels of metals in contaminated habitats, because their diet mainly consists of detritivores (i.e., earthworms) and other soil invertebrates in direct contact with the soil (Beyer et al., 1985).

8.1.4.5.3 Physical/Environmental Factors

Plants

The uptake and distribution of Pb into higher plants from the soil is affected by various chemical and physical factors including the chemical form of Pb, the presence of other metal ions, soil type, soil pH, cation exchange capacity (CEC), the amount of Fe/Mn-oxide films present, organic matter content, temperature, light, and nutrient availability. A small fraction of Pb in soil may be released to the soil moisture, which is then available to be taken up by plants (U.S. Environmental Protection Agency, 1986a).

The form of Pb has an influence on its toxicity to plants. For example, Pb-oxide is less toxic than more bioavailable forms such as Pb-chloride or Pb-acetate. In a study by Khan and Frankland (1983), radish plants were exposed to Pb-oxide and Pb-chloride in a loamy sand at pH 5.4, in a 42-day study. In a tested concentration range of 0 to 5000 mg/kg, root growth was

inhibited by 24% at 500 mg/kg for Pb-chloride and an EC_{50} of 2400 mg/kg was calculated from a dose-response curve. Plant growth ceased at 5000 mg/kg and shoots exhibited an EC_{50} of 2800 mg/kg. For Pb-oxide exposure (concentration range of 0 to 10,000 mg/kg), reported results indicate an EC_{50} of 12,000 mg/kg for shoot growth and an EC_{50} of 10,000 mg/kg for root growth. There was no effect on root growth at 500 mg/kg and a 26% reduction at 1000 mg/kg Pb oxide.

Soil pH is the most influential soil property with respect to uptake and accumulation of Pb into plant species. This is most likely due to increased bioavailability of Pb created by low soil pH. At low soil pH conditions, markedly elevated Pb toxicity was reported for red spruce (*P. rubens*) (Seiler and Paganelli, 1987). At a soil pH of 4.5, ryegrass (*Lolium hybridum*) and oats (*Avena sativa*) had significantly higher Pb concentrations after 3 months of growth compared to plants grown at pH 6.4 (Allinson and Dzialo, 1981).

Invertebrates

The uptake of Pb into invertebrates depends on the physical environment and parameters such as pH, calcium concentration, organic matter content, and CEC. Greater accumulation is found generally when the soil pH or organic content is lower (U.S. Environmental Protection Agency, 1986a).

Soil pH has a significant influence on uptake of Pb into invertebrates. Perämäki et al. (1992) studied the influence of soil pH on uptake into the earthworm *Aporrectodea caliginosa*. Lead accumulation was lowest at the highest pH values, but there was no statistical difference due to variability in the data. Variability in the response also was found by Bengtsson et al. (1986), who reared earthworms (*Dendrobaena rubida*) in acidified soils at pH 4.5, 5.5, or 6.5. Lead uptake into worms was pH-dependent, although the highest concentrations were not always found at the lowest pH. There was no clear relationship between Pb concentration in cocoons and soil pH, and Pb concentrations were higher in the hatchlings than in the cocoons. As has been reported in many other studies (Neuhauser et al., 1995), concentration factors (ratio of Pb in worm to Pb in soil) were lower at higher Pb concentrations in soil. The authors attribute some of this to a lowered feeding activity in worms at higher Pb concentrations (Bengtsson et al., 1986).

Beyer et al. (1987) and Morgan and Morgan (1988) recognized that other factors beyond soil pH could influence the uptake of Pb into earthworms, which may be the cause of the inconsistencies reported by several authors. Both studies evaluated worm uptake of Pb relative

to pH, soil calcium concentration, and organic matter content. Morgan and Morgan (1988) also considered CEC, and Beyer et al. (1987) considered concentrations of phosphorus, potassium, or magnesium in soil. Both studies found that calcium concentrations in soil were correlated with soil pH. Morgan and Morgan (1988) also found that CEC was correlated with percentage organic matter. Soil pH (coupled with CEC) and soil calcium were found to play significant roles in the uptake of Pb into worms (Beyer et al., 1987; Morgan and Morgan, 1988). Beyer et al. (1987) noted that concentrations of phosphorus in soil had no effect.

8.1.4.5.4 Nutritional Factors

Diet is a significant modifier of Pb absorption and of toxic effects in many species of birds and mammals (Eisler, 1988). Dietary deficiencies in calcium, zinc, iron, vitamin E, copper, thiamin, phosphorus, magnesium, fat, protein, minerals, and ascorbic acid increased Pb absorption and its toxic effects (Eisler, 1988).

Mateo et al. (2003b) studied intraspecies sensitivity to Pb-induced oxidative stress, by varying the vitamin E content of mallard diets. Vitamin E can protect against peroxidative damage and was found to decrease the lipid peroxidation in nerves of birds; however, it did not alleviate any sign of the Pb poisoning. The authors hypothesize that inhibition of antioxidant enzymes and interaction with sulfhydryl groups of proteins may have a greater influence on Pb toxicity than lipid peroxidation (Mateo et al., 2003b). The effects observed by Mateo et al. (2003b) were associated with very high concentrations of Pb in diet (1840 mg Pb/kg diet), much higher than would be found generally in the environment, and high enough that the birds decreased their food intake.

Mallard ducklings were exposed to Pb-contaminated sediment and either a low nutrition or optimal nutrition diet (Douglas-Stroebel et al., 2005). Lead exposure combined with a nutritionally-inferior diet caused more changes in behavior (as measured by time bathing, resting, and feeding) than Pb exposure or low-nutrition diet alone. These effects may be due to the low-nutrition diet being deficient in levels of protein, amino acids, calcium, zinc, and other nutrients.

Zebra finches (*Taeniopygia guttata*) were exposed to Pb-acetate via drinking water at 20 mg/L for 38 days, along with either a low- or high-calcium diet (Snoeijs et al., 2005). Lead uptake into tissues was enhanced by a low-calcium diet. Lead did not affect body weight,

hematocrit, or adrenal stress response. Lead suppressed the humoral immune response only in females on a low-calcium diet, suggesting increased susceptibility of females to Pb (Snoeijs et al., 2005).

8.1.4.5.5 Interactions with Other Pollutants

Lead can interact with other pollutants to exert toxicity in an antagonistic (less than additive), independent, additive, or synergistic (more than additive) manner. Concurrent exposure to Pb and additional pollutant(s) can affect the ability of plants to uptake Pb or the other pollutant. However, the uptake and toxic response of plants, exposed to Pb combined with other metals, is inconsistent (Påhlsson, 1989). Therefore, no generalizations can be made about the relative toxicity of metal mixtures. For example, An et al. (2004) conducted acute, 5-day bioassays on cucumber exposed to Pb, Pb + copper, Pb + cadmium, or Pb + copper + cadmium in a sandy loam soil of pH 4.3. Shoot and root growth were measured. Depending on the tissue and metal combination, additivity, synergism, or antagonism was observed in the responses to these metals. In fact, the response in roots was not consistent with the response in shoots for the binary mixtures. However, the combined effects were greater in the roots than the shoots, which may be explained by the tendency for Pb and other heavy metals to be retained in the roots of plants. In addition, the pattern of metal bioaccumulation into plant tissue did not always correlate with the toxic response. However, antagonism was observed in the response of roots and shoots exposed to all three metals, and this was reflected in the decreased accumulation of metals into plant tissues. The authors hypothesized that this may be due to the formation of less bioavailable metal complexes (An et al., 2004).

He et al. (2004) found that selenium and zinc both inhibited the uptake of Pb into Chinese cabbage (*Brassica rapa*) and lettuce (*Lactuca sativa*). Zinc applied at 100 mg/kg or selenium applied at 1 mg/kg decreased the uptake of Pb (present in soil at 10 mg/kg as Pb-nitrate) into lettuce by 15% and 20%, respectively, and into Chinese cabbage by 23 and 20%, respectively.

Selenium compounds were evaluated to determine whether they could change the inhibition of ALAD in liver, kidney, or brain of mice exposed to Pb-acetate (Perottoni et al., 2005). Selenium did not affect the inhibition of ALAD in the kidney or liver, but it did reverse the ALAD inhibition in mouse brain.

Co-occurrence of cadmium with Pb resulted in reduced blood Pb concentrations in rats (Garcia and Corredor, 2004). The authors hypothesized that cadmium may block or antagonize the intestinal absorption of Pb, or the metallothionein induced by cadmium may sequester Pb. However, this was not observed in pigs, where blood Pb concentrations were greater when cadmium was also administered (Phillips et al., 2003). The effect on growth rate also was additive when both metals were given to young pigs (Phillips et al., 2003).

8.1.4.6 Summary

The current document expands upon and updates knowledge related to the uptake, detoxification, physiological effects, and modifying factors of Pb toxicity to terrestrial organisms.

Surface Deposition onto Plants

Recent work (Dalenberg and Van Driel, 1990; Jones and Johnston, 1991; Angelova et al., 2004) has supported previous results and conclusions that surface deposition of Pb onto above-ground vegetation from airborne sources may be significant (U.S. Environmental Protection Agency, 1986a). Similarly, it has been well documented previously that Pb in soil also is taken up by plants, although most remains in the roots, there is little translocation to shoots, leaves, or other plant parts (U.S. Environmental Protection Agency, 1986a). More recent work continues to support this finding (Sieghardt, 1990), and one study found increased tolerance in species with bulbs, possibly due to the storage of Pb in the bulb (Wierzbicka, 1999).

Uptake Mechanism into Plants

Lead was thought previously to be taken up by plants via the symplastic route (through cell membranes), although it was unknown whether some Pb also may be taken up via the apoplastic route (between cells) (U.S. Environmental Protection Agency, 1986a). Recent work has shown that the apoplastic route of transport is stopped in the primary roots by the endodermis (Sieghardt, 1990), supporting the previous conclusion that the symplastic route is the most significant route of tranport into plant cells.

Species Differences in Uptake into Earthworms

Different species of earthworm accumulated different amounts of Pb, and this was not related to feeding strategy (U.S. Environmental Protection Agency, 1986a). This is supported by recent work, which has shown *Aporrectodea* accumulated more than *Lumbricus* (Terhivuo et al., 1994; Pižl and Josens, 1995), although this is not consistently observed (Spurgeon and Hopkin, 1996a).

Speciation and Form of Lead

Recent work supports previous conclusions that the form of metal tested, and its speciation in soil, influence uptake and toxicity to plants and invertebrates (U.S. Environmental Protection Agency, 1986a). The oxide form is less toxic that the chloride or acetate forms, which are less toxic that the nitrate form of Pb (Khan and Frankland, 1983; Lock and Janssen, 2002; Bongers et al., 2004). However, these results must be interpreted with caution, as the counterion (e.g., the nitrate ion) may be contributing to the observed toxicity (Bongers et al., 2004).

Detoxification in Plants

Lead may be deposited in root cell walls as a detoxification mechanism (U.S. Environmental Protection Agency, 1986a), and this may be influenced by calcium concentrations (Antosiewicz, 2005). Yang et al. (2000) suggested that the oxalate content in root and root exudates reduced the bioavailability of Pb in soil, and that this was an important tolerance mechanism. Other hypotheses put forward recently include the presence of sulfur ligands (Sharma et al., 2004) and the sequestration of Pb in old leaves (Szarek-Lukaszewska et al., 2004) as detoxification mechanisms.

Detoxification in Invertebrates

Lead detoxification has not been studied extensively in invertebrates. Glutathione detoxification enzymes were measured in two species of spider (Wilczek et al., 2004). Lead may be stored in waste nodules in earthworms (Hopkin, 1989) or as pyromorphite in the nematode (Jackson et al., 2005).

Physiological Effects

The effects on heme synthesis (as measured by ALAD activity and protoporphyrin concentration, primarily) have been well-documented (U.S. Environmental Protection Agency, 1986a) and continue to be studied (Schlick et al., 1983; Scheuhammer, 1989; Henny et al., 1991; Redig et al., 1991; Beyer et al., 2000; Hoffman et al., 2000a,b). However, Henny et al. (1991) caution that changes in ALAD and other enzyme parameters are not always related to adverse effects, but simply indicate exposure. Other effects on plasma enzymes, which may damage other organs, have been reported (Brar et al., 1997a,b). Lead also may cause lipid peroxidation (Mateo and Hoffman, 2001), which may be alleviated by vitamin E, although Pb poisoning may still result (Mateo et al., 2003b). Changes in fatty acid production have been reported, which may influence immune response and bone formation (Mateo et al., 2003a).

Response Modification

Genetics, biological factors, physical/environmental factors, nutritional factors, and other pollutants can modify terrestrial organism response to Pb. Fisher 344 rats were found to be more sensitive to Pb than Sprague-Dawley rats (Dearth et al., 2004). Younger animals are more sensitive than older animals (Eisler, 1988; Scheuhammer, 1991), and females generally are more sensitive than males (Scheuhammer, 1987; Tejedor and Gonzalez, 1992; Snoeijs et al., 2005). Monogastric animals are more sensitive than ruminants (Humphreys, 1991). Insectivorous mammals may be more exposed to Pb than herbivores (Beyer et al., 1985; Sample et al., 1998), and higher tropic-level consumers may be less exposed than lower trophic-level organisms (Henny et al., 1991). Nutritionally-deficient diets (including low calcium) cause increased uptake of Pb (Snoeijs et al., 2005) and greater toxicity (Douglas-Stroebel et al., 2005) in birds.

Mycorrhizal fungi may ameliorate Pb toxicity until a threshold is surpassed (Malcova and Gryndler, 2003), which may explain why some studies show increased uptake into plants (Lin et al., 2004) while others show no difference or less uptake (Dixon, 1988). Lower soil pH generally increases uptake of Pb into plants and soil invertebrates. However, calcium content, organic matter content, and cation exchange capacity of soils also have had a significant influence on uptake of Pb into plants and invertebrates (Beyer et al., 1987; Morgan and Morgan, 1988).

Interactions of Pb with other metals are inconsistent, depending on the endpoint measured, the tissue analyzed, the animal species, and the metal combination (Phillips et al., 2003; An et al., 2004; He et al., 2004; Garcia and Corredor, 2004; Perottoni et al., 2005).

8.1.5 Exposure-Response of Terrestrial Species

8.1.5.1 Introduction

Section 8.1.4 summarized the most important factors related to uptake of Pb by terrestrial organisms, the physiological effects of Pb, and the factors that modify terrestrial organism responses to Pb. Section 8.1.5 outlines and highlights the critical recent advancements in the understanding of the toxicity of Pb to terrestrial organisms. This section begins with a summary of the conclusions from the 1986 Pb AQCD and then summarizes the critical research conducted on effects of Pb on primary producers, consumers, and decomposers. All concentrations are expressed as mg Pb/kg soil dw, unless otherwise indicated.

The summary of recent critical advancements in understanding toxicity relies heavily on the work completed by a multi-stakeholder group, consisting of federal, state, consulting, industry, and academic participants, led by the EPA to develop Ecological Soil Screening Levels (Eco-SSLs). Eco-SSLs describe the concentrations of contaminants in soils that are protective of ecological receptors (U.S. Environmental Protection Agency, 2005b). They were developed to identify contaminants requiring further evaluation in an ecological risk assessment and were not designed to be used as cleanup levels. Eco-SSLs were derived for terrestrial plants, soil invertebrates, birds, and mammals. Detailed procedures using an extensive list of acceptability and exclusion criteria (U.S. Environmental Protection Agency, 2005b) were used in screening the toxicity studies to ensure that only those that met minimum quality standards were used to develop the Eco-SSLs. In addition, two peer reviews were completed during the Eco-SSL development process. The first was a consultation with the EPA Science Advisory Board (SAB) in April 1999, and the second was a peer review workshop in July 2000, which was open to the public.

Areas of research that were not addressed are effects from irrelevant exposure conditions relative to airborne emissions of Pb (e.g., Pb shot, Pb paint, injection studies, studies conducted on mine tailings, and studies conducted with hydroponic solutions); mixture toxicity (addressed in Section 8.1.4); issues related to indirect effects (e.g., effects on predator/prey interactions,

habitat alteration, etc.); and human health-related research (e.g., hypertension), which is addressed in other sections of this document.

The toxicity data presented herein should be reviewed with a note of caution regarding their relevance to field conditions. Laboratory studies, particularly those using Pb-spiked soil, generally do not allow the soil to equilibrate following the addition of Pb and prior to the addition of test organisms. This may result in increased bioavailability and overestimation of Pb toxicity relative to actual environmental conditions (Davies et al., 2003).

8.1.5.2 Summary of Conclusions from the 1986 Lead Criteria Document

The previous Pb AQCD, Volume II (U.S. Environmental Protection Agency, 1986a) reviewed the literature on the toxicity of Pb to plants, soil organisms, birds, and mammals. The main conclusions from this document are provided below.

Primary Producers

Commonly reported effects of Pb on vascular plants include the inhibition of photosynthesis, respiration, and/or cell elongation, all of which reduce plant growth. However, it was noted that studies of other effects on plant processes such as maintenance, flowering, and hormone development had not been conducted; therefore, no conclusion could be reached concerning effects of Pb on these processes.

The EPA (U.S. Environmental Protection Agency, 1986a) concluded that most plants experience reduced growth when Pb concentrations in soil moisture (the film of moisture surrounding soil particles in the root zone of soil) exceed 2 to 10 mg/kg. It also was concluded that most plants would experience reduced growth (inhibition of photosynthesis, respiration, or cell elongation) in soils of ≥10,000 mg/kg when soil composition and pH are such that bioavailability of Pb in the soil is low (see Section 8.1.4 for details on factors affecting bioavailability of Pb in soil). Acid soils or soils with low organic matter tend to increase Pb bioavailability and would inhibit plants at much lower Pb concentrations (e.g., as low as <100 mg/kg).

Many effect levels have been reported at Pb concentrations much lower than 10,000 mg/kg soil. For example, effects on rye grass (*Lolium rigidum*) exposed to Pb in soil included inhibition of germinating root elongation (at <2.5 mg/kg), absence of root growth

(at 5 mg/kg), or 55% inhibition of seed germination (at 20 to 40 mg/kg). Stunted growth in radish (*Raphanus sativus*) was observed at 1000 mg/kg soil, with complete growth inhibition at 5000 mg/kg, when Pb was added as Pb-chloride; effects were less severe when the Pb was added as Pb-oxide.

Consumers

The EPA (U.S. Environmental Protection Agency, 1986a) concluded that food is the largest contributor of Pb to animals, with inhalation rarely accounting for more than 10 to 15% of daily intake of Pb and drinking water exposures being quite low. It also was concluded that a regular dose of 2 to 8 mg/kg-day causes death in most animals. Grazing animals may consume more than 1 mg/kg-day in habitats near smelters and roadsides, but no toxic effects were documented in these animals.

Decomposers

Lack of decomposition has been observed as a particular problem around smelter sites. Lead concentrations between 10,000 and 40,000 mg/kg soil can eliminate populations of decomposer bacteria and fungi (U.S. Environmental Protection Agency, 1986a). Lead may affect decomposition processes by direct toxicity to specific groups of decomposers, by deactivating enzymes excreted by decomposers to break down organic matter, or by binding with the organic matter and rendering it resistant to the action of decomposers.

Microorganisms are more sensitive than plants to Pb in soil. Delayed decomposition may occur at between 750 and 7500 mg/kg soil (depending on soil type and other conditions). Nitrification is inhibited by 14% at 1000 mg/kg soil.

U.S. Environmental Protection Agency Staff Review of 1986 Criteria Document

The EPA reviewed the 1986 Pb AQCD and presented an overall summary of conclusions and recommendations (U.S. Environmental Protection Agency, 1990). The major conclusion was that available laboratory and field data indicated that high concentrations of Pb can affect certain plants and alter the composition of soil microbial communities. It was noted that few field studies were available in which Pb exposures and associated effects in wildlife were reported.

8.1.5.3 Recent Studies on the Effects of Lead on Primary Producers

Several studies published since 1986 have reports of terrestrial plant exposure to Pb in soil, many of which were reviewed during the development of the Eco-SSLs (U.S. Environmental Protection Agency, 2005a). The relevant information from the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) is summarized below. A literature search and review also was conducted to identify critical papers published since 2002, which is when the literature search was completed for Eco-SSL development, and no new papers were identified as critical to the understanding of Pb toxicity to terrestrial primary producers.

Effects observed in studies conducted since the 1986 Pb AQCD are similar to those reported previously and include decreased photosynthetic and transpiration rates and decreased growth and yield (U.S. Environmental Protection Agency, 2005a). The phytotoxicity of Pb is considered relatively low, due to the limited availability and uptake of Pb from soil and soil solution and minimal translocation of Pb from roots to shoots (Påhlsson, 1989). Although many laboratory toxicity studies have reported effects on plants, there are few reports of phytotoxicity from Pb exposure under field conditions. For example, Leita et al. (1989) and Sieghardt (1990) reported high concentrations of Pb and other metals in soil and vegetation collected around mining areas in Europe, with no toxicity symptoms observed in plants or fruit.

The literature search completed for the terrestrial plant Eco-SSL development identified 439 papers for detailed review, of which 28 met the minimum criteria (U.S. Environmental Protection Agency, 2005b). Thirty ecotoxicological endpoints were gleaned from these 28 papers and were further evaluated; most of those evaluated growth (biomass), which was considered the most sensitive and ecologically-relevant endpoint (U.S. Environmental Protection Agency, 2005a). Five of the endpoints, representing four species tested under three different combinations of pH and organic matter content, were used to develop the Eco-SSL of 120 mg/kg (115 mg/kg rounded to two significant digits) (Table 8-1.5.1).

The 25 ecotoxicological endpoints that were not used to develop the Eco-SSL for plants are presented in Table 8-1.5.2. The first six endpoints were considered eligible for Eco-SSL derivation but were not used; the remainder did not meet all of the requirements to be considered for inclusion in the Eco-SSL derivation process.

Table 8-1.5.1. Plant Toxicity Data Used to Develop the Eco-SSL

Plant Species	Soil pH	% Organic Matter	Toxicity Parameter	Pb in Soil (mg/kg dw)
Loblolly pine (*Pinus taeda*)	4	2.5	MATC* (growth)	144
Red maple (*Acer rubrum*)	4	2.5	MATC (growth)	144
Berseem clover (*Trifolium alexandrium*)	6.3	0.94	MATC (growth)	316
Berseem clover	6.7	3.11	MATC (growth)	141
Rye grass (*Lolium rigidum*)	5.6	0.1	MATC (growth)	22
			Geometric Mean	115

*MATC = Maximum Acceptable Threshold Concentration, or the geometric mean of the NOAEC (no-observed-adverse-effect concentration) and LOAEC (lowest-observed-adverse-effect concentration).

Source: U.S. Environmental Protection Agency (2005a).

8.1.5.4 Recent Studies on the Effects of Lead on Consumers

Since the 1986 Pb AQCD, there have been several studies in which birds and mammals were exposed to Pb via ingestion (primarily through dietary Pb). Many of these were reviewed during development of the Eco-SSLs (U.S. Environmental Protection Agency, 2005a). The relevant information from the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) is described below. A literature search and review was conducted to identify critical papers published since 2002. These recent critical papers are described briefly below. No studies were found that used inhalation exposures to evaluate endpoints such as survival, growth, and reproduction in birds or mammals. All studies described below exposed organisms via ingestion (drinking water or diet) or gavage.

The Eco-SSLs for avian and mammalian consumers are presented as Pb concentrations in soil. These concentrations were calculated by assuming exposure to Pb via incidental soil ingestion and ingestion of Pb-contaminated food, and using a no-observed-adverse-effect level (NOAEL) as the toxicity reference value (TRV) (U.S. Environmental Protection Agency, 2005b). A simplified version of the equation used to calculate the Eco-SSL is:

Table 8-1.5.2. Plant Toxicity Data Not Used to Develop the Eco-SSL

Plant Species	Soil pH	% Organic Matter	Toxicity Parameter	Pb in Soil (mg/kg dw)
Studies eligible for Eco-SSL derivation, but not used				
Berseem clover (*Trifolium alexandrium*)	6.7	3.11	MATC	141
Tomato (*Lycopericon esculentum*)	7.73	1.70	MATC	71
Tomato	8.20	0.86	MATC	71
Fenugreek (*Trigonella foenum-graecum*)	8.3	0.5	MATC	283
Spinach (*Spinacea oleracea*)	6.7	3.0	MATC	424
Corn (*Zea mays*)	6.5	2.1	MATC	158
Sow thistle (*Sonchus oleraceus*)	7.23	1.6	MATC	2,263
Studies not eligible for Eco-SSL derivation				
Loblolly pine (*Pinus taeda*)	5.5	3.4	NOAEC	480
Red oak (*Quercus rubra*)	6	1.5	LOAEC	100
Spinach	6.7	0.0	NOAEC	600
Alfalfa (*Medicago sativa*)	6.4	1.0	NOAEC	250
Alfalfa	6.9	1.7	NOAEC	250
Alfalfa	6.9	1.7	NOAEC	250
Radish (*Raphanus sativus*)	6.9	1.0	LOAEC	500
Radish	6.9	1.0	LOAEC	100
Radish	6.9	1.0	LOAEC	100
Onion (*Allium cepa*)	8.3	0.5	LOAEC	50
Radish	5.1	8.0	NOAEC	600
Carrot (*Daucus carota*)	7.0	0.6	NOAEC	85
Peas (*Pisum sativum*)	7.0	0.6	NOAEC	85
Barley (*Hordeum vulgare*)	6.0	2.5	NOAEC	1,000
Alfalfa	6.9	4.8	NOAEC	250
Tomato	7.45	2.06	MATC	35
Spinach	6.7	8.0	NOAEC	600
Radish	6.2	8.0	NOAEC	600
Radish	7.1	8.0	NOAEC	600

*MATC = Maximum Acceptable Threshold Concentration, or the geometric mean of the NOAEC (no-observed-adverse-effect concentration) and LOAEC (lowest-observed-adverse-effect concentration).

Source: U.S. Environmental Protection Agency (2005a).

$$HQ = \frac{[(C_{soil} \times IR_{soil}) + (C_{food} \times IR_{food})] / BW}{TRV} \quad (8\text{-}4)$$

where:

HQ = hazard quotient (1 mg Pb/kg bw/day)
C_{soil} = concentration of Pb in soil (mg Pb/kg soil)
IR_{soil} = incidental soil ingestion rate (kg soil/day)
C_{food} = concentration of Pb in food (mg Pb/kg food)
IR_{food} = food ingestion rate (kg food/day)
BW = body weight (kg)
TRV = toxicity reference value (mg Pb/kg bw/day)

Food ingestion was estimated by modeling the uptake of Pb from soil into each diet component (e.g., vegetation, invertebrates, etc.). Bioavailability of Pb in soil and food was assumed to be 100%. The Eco-SSL is equivalent to the concentration of Pb in soil that results in an HQ = 1. The two factors that may have the most significant influence on the resulting Eco-SSL are the assumption of 100% bioavailability of Pb in soil and diet and the selection of the TRV. The toxicity data that were reviewed to develop the TRV are presented in the following subsections.

Representative avian and mammalian wildlife species were selected for modeling Pb exposures to wildlife with different diets and calculating the Eco-SSL. The avian species selected were dove (herbivore), woodcock (insectivore), and hawk (carnivore). The mammalian species selected were vole (herbivore), shrew (insectivore), and weasel (carnivore). The lowest of the three back-calculated soil concentrations, which resulted in an HQ = 1, was selected as the Eco-SSL. For Pb, the lowest values were for the insectivorous species of bird and mammal.

Avian Consumers

Effects on birds observed in studies conducted since the 1986 Pb AQCD are similar to those reported previously: mortality, changes in juvenile growth rate and weight gain, effects on various reproductive measures, and changes in behavior (U.S. Environmental Protection Agency, 2005a). Reproductive effects following Pb exposure included declines in clutch size, number of young hatched, and number of young fledged as well as decreased fertility or eggshell thickness.

Few significant reproductive effects have been reported in birds at Pb concentrations below 100 mg/kg in the diet (Scheuhammer, 1987).

The literature search completed for Eco-SSL development identified 2,429 papers for detailed review for either avian or mammalian species, of which 54 met the minimum criteria for further consideration for avian Eco-SSL development (U.S. Environmental Protection Agency, 2005a). The 106 toxicological data points for birds that were further evaluated included biochemical, behavioral, physiological, pathological, reproductive, growth, and survival effects. Growth and reproduction data were used to derive the Eco-SSL (Table 8-1.5.3; Figure 8-1.5.1), as these were determined to be the most ecologically-relevant endpoints. The geometric mean of the NOAELs was calculated as 10.9 mg/kg-day, which was higher than the lowest bounded LOAEL (the term "bounded" means that both a NOAEL and LOAEL were obtained from the same study). Therefore, the highest bounded NOAEL that was lower than the lowest bounded LOAEL for survival, growth, or reproduction (1.63 mg Pb/kg bw-day) was used as the TRV. The TRV was used to back-calculate the Eco-SSL of 11 mg/kg soil for avian species.

Many of the toxicity data presented in the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) are lower than those discussed in the 1986 Pb AQCD. The TRV and resulting Eco-SSL were derived using many conservative assumptions. For example, the EPA (U.S. Environmental Protection Agency, 2005a) recognizes that toxicity is observed over a wide range of doses (<1 to >100 mg Pb/kg bw/day), even when considering only reproductive effects in the same species. In addition, the TRV of 1.63 mg/kg-day is lower than most of the reported doses that have been associated with measured effects. This is true not only for survival, growth, and reproductive effects but also biochemical, behavioral, physiological, and pathological effects, which generally are observed at lower concentrations than effects on growth or reproduction. In addition, the Eco-SSL was back-calculated using conservative modeling assumptions. Therefore, the Eco-SSL of 11 mg/kg may be considered a conservative value.

Very little research has been done to expand the knowledge of the toxicity of Pb to birds since the Eco-SSL work was done. However, several studies have been conducted on waterfowl. Toxicity data for waterfowl (in particular, mallards) were included in the soil Eco-SSL development process (Table 8-1.5.3), although mallards may be more exposed to contaminants in sediment than soil. Effects on waterfowl may vary depending on the form of Pb, characteristics of the sediment, the foraging strategy of the species (which may vary during

Table 8-1.5.3. Avian Toxicity Data Used to Develop the Eco-SSL

Avian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Reproduction													
Japanese quail	4	FD	5	w	6	w	LB	F	REP	PROG	WO	0.194	1.94
Chicken	3	FD	4	w	NR	NR	LB	F	REP	PROG	WO	1.63	3.26
Chicken	4	FD	30	d	22	w	LB	F	EGG	ALWT	EG	2.69	4.04
Mallard	2	FD	76	d	NR	NR	SM	F	EGG	ESTH	EG	5.63	
American kestrel	3	FD	6	mo	1-6	yr	AD	F	REP	RSUC	WO	12.0	
Japanese quail	5	FD	5	w	6	d	JV	M	REP	TEWT	TE	12.6	126
Japanese quail	5	FD	5	w	1	d	JV	M	REP	TEWT	TE	67.4	135
Japanese quail	3	FD	32	d	NR	NR	AD	F	REP	PROG	WO	125	
Japanese quail	5	FD	12	w	0	d	LB	B	REP	EGPN	EG		0.110
Japanese quail	4	FD	12	w	NR	NR	LB	F	REP	PROG	WO		0.194
Chicken	5	FD	10	w	NR	NR	LB	F	REP	PROG	WO		3.26
Ringed turtle dove	2	DR	11	w	NR	NR	AD	M	REP	TEWT	TE		11.8
Japanese quail	2	FD	1	w	14	w	JV	F	REP	TPRD	WO		93.1
Japanese quail	2	FD	27	d	NR	NR	AD	F	REP	PROG	WO		377

Table 8-1.5.3 (cont'd). Avian Toxicity Data Used to Develop the Eco-SSL

Avian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Growth													
Japanese quail	3	FD	5	w	1	d	JV	F	GRO	BDWT	WO	1.56	15.6
Japanese quail	3	FD	2	w	1	d	JV	B	GRO	BDWT	WO	2.77	
Japanese quail	2	FD	2	w	1	d	JV	NR	GRO	BDWT	WO	4.64	
Japanese quail	3	FD	4	w	0	d	JV	F	GRO	BDWT	WO	5.93	59.3
Chicken	4	FD	4	w	4	w	JV	NR	GRO	BDWT	WO	6.14	61.4
Chicken	4	FD	4	w	4	w	JV	NR	GRO	BDWT	WO	7.10	71.0
Japanese quail	5	FD	12	w	0	d	JV	F	GRO	BDWT	WO	11.1	111
Japanese quail	5	FD	12	w	1	w	JV	F	GRO	BDWT	WO	11.2	112
Japanese quail	5	FD	2	w	6	d	JV	NR	GRO	BDWT	WO	12.6	126
Japanese quail	5	FD	1	w	1	d	JV	NR	GRO	BDWT	WO	13.5	67.4
Chicken	2	FD	21	d	1	d	JV	B	GRO	BDWT	WO	14.2	
Duck	3	GV	3	mo	24	w	MA	F	GRO	BDWT	WO	20.0	
American kestrel	4	GV	10	d	1	d	JV	NR	GRO	BDWT	WO	25.0	125
Chicken	2	FD	20	d	1	d	JV	B	GRO	BDWT	WO	28.4	
Japanese quail	5	FD	14	d	1	d	JV	B	GRO	BDWT	WO	34.5	

Table 8-1.5.3 (cont'd). Avian Toxicity Data Used to Develop the Eco-SSL

Avian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
American kestrel	4	FD	60	d	1-2	yr	AD	B	GRO	BDWT	WO	54.3	
Chicken	5	FD	2	w	1	d	JV	M	GRO	BDWT	WO	61.3	123
Mallard	4	FD	8	d	9	d	JV	NR	GRO	BDWT	WO	66.9	
Chicken	5	FD	20	d	1	d	JV	M	GRO	BDWT	WO		38.2
Chicken	2	FD	3	w	1	d	JV	M	GRO	BDWT	WO		53.1
Japanese quail	3	FD	32	d	NR	NR	AD	F	GRO	BDWT	WO		64.3
Chicken	2	FD	19	d	1	d	JV	M	GRO	BDWT	WO		76.3
Chicken	3	FD	2	w	1	d	JV	M	GRO	BDWT	WO		124
Chicken	4	FD	14	d	8	d	JV	M	GRO	BDWT	WO		152
Chicken	2	FD	20	d	1	d	JV	M	GRO	BDWT	WO		163
Chicken	2	OR	4	w	NR	NR	JV	B	GRO	BDWT	WO		200
Chicken	2	FD	7	d	1	d	JV	M	GRO	BDWT	WO		262
Chicken	2	FD	2	w	1	d	JV	M	GRO	BDWT	WO		270
Chicken	2	FD	7	d	1	d	IM	NR	GRO	BDWT	WO		273
Chicken	2	FD	14	d	8	d	JV	M	GRO	BDWT	WO		282

AD = adult; ALWT = albumin weight; B = both; BDWT = body weight changes; d = days; DR = drinking water; EG = egg; EGG = effects on eggs; EGPN = egg production; ESTH = eggshell thinning; F = female; FD = food; GRO = growth; GV = gavage; JV = juvenile; LB = laying bird; MA = mature; M = male; mo = months; NR = not reported; OR = other oral; PROG = progeny counts or numbers; REP = reproduction; RSUC = reproductive success; SM = sexually mature; TE = testes; TEWT = testes weight; TPRD = total production; w = weeks; WO = whole organism; yr = years.

Source: U.S. Environmental Protection Agency (2005a)

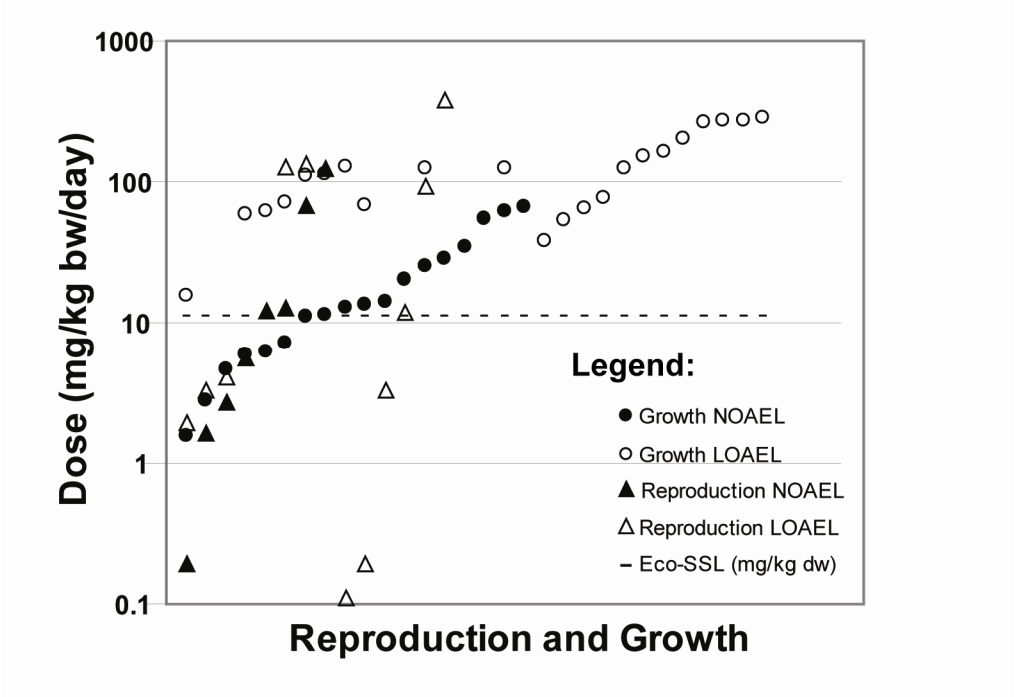

Figure 8-1.5.1. Avian toxicity data considered in development of the Eco-SSL.

Source: U.S. Environmental Protection Agency (2005a).

reproduction), and the nutritional status of the animal. Sediment is recognized as an important route of exposure for waterfowl, particularly those species that dabble (i.e., forage on invertebrates in the sediment) (Beyer et al., 2000; Douglas-Stroebel et al., 2005). Douglas-Stroebel et al. (2005) found that mallard ducklings exposed to Pb-contaminated sediment and a low nutrition diet exhibited more changes in behavior (as measured by time bathing, resting, and feeding) than Pb exposure or low nutrition exposure alone. These effects may be due to the low nutrition diet being deficient in levels of protein, amino acids, calcium, zinc, and other nutrients.

Beyer et al. (2000) related blood Pb to sublethal effects in waterfowl along the Coeur d'Alene River near a mining site in Idaho. The authors suggested that 0.20 mg/kg ww blood Pb represents the no-effect level. This no-effect blood concentration corresponds to a sediment Pb concentration of 24 mg/kg. A sediment concentration of 530 mg/kg, associated with a blood Pb concentration of 0.68 mg/kg ww, is suggested to be the lowest-effect concentration. These results are consistent with those of Scheuhammer (1989) who found blood Pb concentrations of 0.18 µg/mL to 0.65 µg/mL in mallards corresponded to conditions associated with greater than

normal exposure to Pb, but that that should not be considered Pb poisoning. The study by Beyer et al. (2000) related blood Pb to waterfowl mortality and concluded that some swan mortality may occur at blood Pb levels of 1.9 mg/kg ww, corresponding to a sediment Pb concentration of 1800 mg/kg. Using the mean blood level of 3.6 mg/kg ww from all moribund swans in the study, it was predicted that half of the swans consuming sediment at the 90th percentile rate would die with chronic exposure to sediment concentrations of 3600 mg/kg.

Mammalian Consumers

Effects on mammals observed in studies conducted since the 1986 AQCD are similar to those reported previously: mortality, effects on reproduction, developmental effects, and changes in growth (U.S. Environmental Protection Agency, 2005a). Very little research has been done to expand the knowledge of the toxicity of Pb to mammalian wildlife, since the Eco-SSL work was done. Most studies conducted on mammals use laboratory animals to study potential adverse effects of concern for humans, and such studies are summarized in other sections of this document.

Of the 2,429 papers identified in the literature search for Eco-SSL development, 219 met the minimum criteria for further consideration for mammalian Eco-SSL development (U.S. Environmental Protection Agency, 2005a). The 343 ecotoxicological endpoints for mammals that were further evaluated included biochemical, behavioral, physiological, pathological, reproductive, growth, and survival effects. Growth and reproduction data were used to derive the Eco-SSL (Table 8-1.5.4, Figure 8-1.5.2), as these were determined to be the most ecologically-relevant endpoints. The geometric mean of the NOAELs was calculated as 40.7 mg/kg-day, which was higher than the lowest bounded LOAEL for survival, growth or reproduction. Therefore, the highest bounded NOAEL that was lower than the lowest bounded LOAEL for survival, growth, or reproduction (4.7 mg Pb/kg bw-day) was used as the TRV. The TRV was used to back-calculate the Eco-SSL of 56 mg/kg soil.

A review of the data presented in the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) reveals that effects on survival generally are observed at Pb doses much greater than those reported in the 1986 Pb AQCD, where it was concluded that most animals would die when consuming a regular dose of 2 to 8 mg Pb/kg bw-day (U.S. Environmental Protection Agency, 1986a). However, the data presented in the Eco-SSL document (U.S. Environmental

Table 8-1.5.4. Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Reproduction													
Rat	5	DR	62	d	21	d	GE	F	REP	PRWT	WO	0.71	7.00
Rat	6	DR	21	d	NR	NR	GE	F	REP	PRWT	WO	1.00	5.00
Rat	3	DR	35	d	NR	NR	AD	M	REP	RSUC	WO	2.60	26.0
Rat	4	DR	62	d	21	d	GE	B	REP	PRWT	WO	3.00	6.0
Sheep	3	FD	27	w	NR	NR	GE	F	REP	RSUC	WO	4.50	—
Rat	6	DR	21	d	NR	NR	GE	F	REP	PRWT	WO	5.00	10.0
Guinea pig	3	DR	40	d	NR	NR	GE	F	REP	PRWT	WO	5.50	—
Rat	5	FD	92	w	21	d	JV	M	REP	TEWT	TE	7.50	74.9
Rat	4	DR	23.8	d	21	d	LC	F	REP	Other	WO	8.90	
Rat	5	DR	23.8	d	21	d	GE	F	REP	Other	WO	9.10	45.0
Cotton rat	3	DR	7	w	NR	NR	AD	M	REP	RHIS	RT	12.4	170
Rat	4	GV	9	w	10	w	JV	M	REP	SPCV	TE	18.0	180
Rat	3	DR	100	d	21	d	GE	F	REP	PRWT	WO	25.4	—
Rat	2	FD	35	d	70	d	LC	F	REP	PRWT	WO	27.5	—
Rat	4	DR	60	d	NR	NR	SM	M	REP	TEWT	TE	31.6	63.2
Rat	4	DR	56	d	70	d	LC	F	REP	PROG	WO	32.5	—
Rat	3	DR	31	d	NR	NR	LC	F	REP	PRWT	WO	33.3	111
Rat	4	GV	41	d	NR	NR	GE	F	REP	PRWT	WO	41.0	54.6
Rat	5	DR	1	w	94	d	JV	M	REP	SPCL	SM	47.3	82.0
Rat	4	DR	30	d	NR	NR	SM	M	REP	Other	SV	56.0	285

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Hamster	2	DR	51	d	15	w	GE	F	REP	PROG	WO	64.8	—
Hamster	2	DR	14	d	11	w	GE	F	REP	PROG	WO	64.9	—
Rat	4	DR	37	d	NR	NR	GE	F	REP	PRWT	WO	90.1	270
Rat	5	GV	12	d	NR	NR	GE	F	REP	RSEM	EM	100	150
Rat	3	DR	68	d	25	d	GE	F	REP	PRWT	WO	115	—
Rat	4	DR	77	d	25	d	GE	F	REP	PRWT	WO	116	—
Rat	2	DR	21	d	NR	NR	LC	F	REP	PRWT	WO	120	—
Mouse	3	FD	8	w	2	mo	GE	M	REP	SPCV	TE	144	1,440
Mouse	7	FD	30	d	NR	NR	LC	F	REP	PRWT	WO	202	506
Mouse	7	FD	30	d	NR	NR	LC	F	REP	PRWT	WO	202	506
Rat	4	DR	21	d	NR	NR	GE	F	REP	DEYO	WO	276	552
Rat	5	DR	10	w	NR	NR	AD	M	REP	TEWT	MT	294	587
Rat	2	GV	102	d	30	d	GE	F	REP	PRWT	WO	441	—
Rat	2	DR	9	mo	NR	NR	SM	M	REP	RHIS	TE	600	—
Rat	4	FD	4	d	NR	NR	LC	F	REP	PRWT	WO	601	1,500
Rat	4	DR	13	w	NR	NR	JV	M	REP	FERT	WO	639	—
Mouse	4	GV	60	d	NR	NR	AD	F	REP	RPRD	OV	—	2.00
Rat	3	FD	339	d	26-27	d	JV	B	REP	PRWT	WO	—	2.49
Rat	2	DR	9	mo	21	d	JV	F	REP	DEYO	WO	—	2.94
Mouse	2	DR	6	mo	21	d	JV	F	REP	DEYO	WO	—	3.62
Mouse	4	GV	52	d	2	mo	GE	F	REP	PROG	EM	—	5.50
Rat	2	DR	120	d	1	d	GE	M	REP	SPCL	TE	—	6.76

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Mouse	2	DR	5	w	NR	NR	AD	M	REP	TEDG	TE	—	16.6
Mouse	2	GV	2	w	NR	NR	JV	M	REP	SPCL	SM	—	46.4
Rat	2	FD	102	d	NR	NR	GE	F	REP	PROG	WO	—	49.6
Rat	2	GV	3	mo	8	w	SM	M	REP	TEDG	TE	—	50.0
Rat	2	DR	18	d	NR	NR	GE	F	REP	PRWT	WO	—	55.5
Rat	3	DR	90	d	NR	NR	AD	M	REP	SPCL	SM	—	61.2
Mouse	2	DR	23	d	NR	NR	GE	F	REP	PRWT	WO	—	78.6
Mouse	2	DR	62	d	NR	NR	GE	F	REP	PRWT	WO	—	99.8
Mouse	2	DR	18	w	6-8	w	LC	F	REP	PRWT	WO	—	137
Mouse	2	DR	12	w	9	w	SM	M	REP	PRFM	WO	—	139
Mouse	4	FD	18	d	NR	NR	GE	F	REP	PRWT	WO	—	154
Rat	2	DR	4	w	99	d	JV	M	REP	SPCL	SM	—	171
Rat	5	DR	6	w	4	mo	GE	F	REP	RHIS	WO	—	175
Rat	2	DR	22	d	NR	NR	GE	F	REP	PRWT	WO	—	178
Rat	3	DR	30	d	52	d	JV	M	REP	GREP	PG	—	198
Rat	2	DR	13	w	NR	NR	GE	F	REP	PRWT	WO	—	200
Rat	2	DR	21	d	80	d	JV	F	REP	PRWT	WO	—	218
Rat	4	FD	3	w	NR	NR	LC	F	REP	PRWT	WO	—	221
Rat	2	FD	1	w	19	w	LC	F	REP	PRWT	WO	—	222
Rat	4	FD	3	w	NR	NR	LC	F	REP	PRWT	WO	—	230
Rat	3	FD	25	d	NR	NR	LC	F	REP	PRWT	WO	—	258
Rat	2	DR	21	d	NR	NR	LC	F	REP	PRWT	WO	—	330

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Rat	2	DR	30	d	52	d	JV	M	REP	SPCL	SM	—	354
Rat	2	DR	17	d	NR	NR	GE	F	REP	PRWT	WO	—	360
Rat	2	DR	24	d	NR	NR	LC	F	REP	PRWT	WO	—	360
Rat	2	DR	12	d	NR	NR	GE	F	REP	PRWT	WO	—	362
Rat	2	DR	30	d	27	d	JV	M	REP	SPCL	SM	—	364
Mouse	2	DR	44	d	NR	NR	GE	F	REP	PRWT	WO	—	381
Mouse	2	DR	14	d	NR	NR	LC	F	REP	PRWT	WO	—	381
Rat	2	DR	50	d	24	d	JV	F	REP	RBEH	WO	—	381
Mouse	2	DR	45	d	50-100	d	GE	F	REP	ODVP	WO	—	404
Rat	2	DR	22	d	NR	NR	GE	F	REP	PRWT	WO	—	420
Mouse	2	DR	48	d	NR	NR	GE	F	REP	PRWT	WO	—	437
Rat	2	DR	9	mo	3	mo	SM	M	REP	SPCL	TE	—	579
Rat	2	DR	9	mo	NR	NR	SM	M	REP	TEDG	TE	—	600
Rat	2	DR	3	w	14	w	LC	F	REP	PRWT	WO	—	635
Mouse	2	FD	7	d	NR	NR	GE	F	REP	RSUC	EM	—	646
Rat	2	DR	126	d	1	d	GE	F	REP	PROG	WO	—	651
Rat	2	DR	20	w	10	w	GE	F	REP	PRWT	WO	—	750
Mouse	2	DR	4	d	NR	NR	LC	F	REP	PRWT	WO	—	762
Rat	2	FD	2	w	NR	NR	LC	F	REP	PRWT	WO	—	828
Rat	2	FD	7	d	NR	NR	LC	F	REP	PRWT	WO	—	833
Rat	2	FD	21	d	NR	NR	LC	F	REP	PRWT	WO	—	991
Mouse	4	DR	18	w	11	w	JV	F	REP	TEWT	WO	—	1,370

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Rat	2	FD	30	d	NR	NR	LC	F	REP	PRWT	WO	—	1,770
Mouse	2	DR	14	w	NR	NR	GE	B	REP	PROG	WO	—	1,990
Rat	2	FD	16	d	NR	NR	LC	F	REP	PROG	WO	—	2,570
Rat	2	FD	7	d	NR	NR	LC	F	REP	PRWT	WO	—	2,570
Rat	2	FD	25	d	NR	NR	LC	F	REP	PRWT	WO	—	2,570
Rat	M	FD	27	d	NR	NR	LC	C	REP	PROG	WO	—	2,840
Mouse	2	DR	14	w	21	d	JV	B	REP	PROG	WO	—	3,630
Rat	2	FD	17	d	NR	NR	LC	F	REP	PRWT	WO	—	6,170
Growth													
Horse	2	FD	15	w	20-21	w	JV	M	GRO	BDWT	WO	0.15	—
Rat	2	FD	21	d	0	d	JV	F	GRO	BDWT	WO	0.5	—
Rat	6	DR	21	d	NR	NR	GE	F	GRO	BDWT	WO	1.00	5.00
Rat	5	DR	7	d	50	d	AD	F	GRO	BDWT	WO	1.27	13.0
Cattle	4	OR	7	w	1	w	JV	M	GRO	BDWT	WO	1.99	—
Rat	3	DR	14	d	21	d	JV	F	GRO	BDWT	WO	2.40	—
Rat	2	DR	332	d	28	d	JV	B	GRO	BDWT	WO	2.98	—
Rat	4	DR	7	w	21	d	GE	F	GRO	BDWT	WO	4.70	8.90
Dog	3	FD	7	mo	NR	NR	JV	NR	GRO	BDWT	WO	4.71	—
Rat	3	DR	30	d	22-24	d	JV	M	GRO	BDWT	WO	5.64	28.2
Rat	4	DR	23	d	22	d	JV	F	GRO	BDWT	WO	5.80	29.0
Cattle	3	OR	84	d	NR	NR	JV	M	GRO	BDWT	WO	7.79	—
Rat	2	OR	6	w	NR	NR	AD	M	GRO	BDWT	WO	9.10	—

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Rat	2	GV	8	w	NR	NR	JV	F	GRO	BDWT	WO	10.0	—
Rat	3	DR	6	mo	NR	NR	AD	M	GRO	BDWT	WO	10.6	532
Rabbit	3	GV	10	d	1	d	JV	F	GRO	BDWT	WO	10.7	50.4
Rat	2	DR	140	d	21	d	JV	M	GRO	BDWT	WO	10.7	—
Rat	2	DR	6	w	NR	NR	JV	M	GRO	BDWT	WO	15.1	—
Rat	2	FD	10	w	NR	NR	JV	M	GRO	BDWT	WO	15.4	—
Rat	2	OR	6	w	NR	NR	AD	M	GRO	BDWT	WO	15.5	—
Rat	2	DR	7	w	NR	NR	JV	M	GRO	BDWT	WO	16.1	—
Mouse	3	DR	14	d	0	d	JV	NR	GRO	BDWT	WO	16.3	163
Rat	4	GV	9	w	10	w	JV	M	GRO	BDWT	WO	18.0	180
Rat	3	FD	339	d	26-27	d	JV	B	GRO	BDWT	WO	18.3	—
Rat	4	GV	29	d	NR	NR	SM	F	GRO	BDWT	WO	18.9	—
Rat	7	DR	10	w	NR	NR	JV	M	GRO	BDWT	WO	24.3	—
Rat	4	DR	56	d	70	d	LC	F	GRO	BDWT	WO	32.5	—
Sheep	5	FD	84	d	NR	NR	JV	M	GRO	BDWT	WO	32.7	—
Rat	2	DR	10	w	NR	NR	JV	M	GRO	BDWT	WO	38.5	—
Cattle	4	FD	7	w	16	w	JV	M	GRO	BDWT	WO	43.0	—
Rat	2	GV	28	d	2	d	JV	B	GRO	BDWT	WO	50.0	—
Rat	5	DR	4	w	94	d	JV	M	GRO	BDWT	WO	71.5	178
Rat	4	GV	12	d	2	d	JV	B	GRO	BDWT	WO	75.0	225
Rat	2	FD	4	w	NR	NR	JV	M	GRO	BDWT	WO	100	—
Rat	6	DR	10	w	NR	NR	JV	M	GRO	BDWT	WO	120	383

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Mouse	3	FD	4	w	3	mo	JV	B	GRO	BDWT	WO	136	1360
Mouse	2	DR	18	w	6-8	w	LC	F	GRO	BDWT	WO	137	—
Mouse	2	DR	12	w	NR	NR	GE	M	GRO	BDWT	WO	139	—
Rat	3	DR	30	d	52	d	JV	M	GRO	BDWT	WO	169	508
Rat	2	DR	4	w	99	d	JV	B	GRO	BDWT	WO	171	—
Rat	4	GV	18	d	3	d	JV	M	GRO	BDWT	WO	180	—
Mouse	3	DR	6	w	7	w	SM	M	GRO	BDWT	WO	187	373
Rat	4	GV	18	d	2	d	JV	B	GRO	BDWT	WO	200	—
Rat	2	GV	91	d	NR	NR	JV	M	GRO	BDWT	WO	200	—
Rat	2	DR	21	d	80	d	JV	F	GRO	BDWT	WO	218	—
Rat	4	FD	1	w	NR	NR	LC	F	GRO	BDWT	WO	230	460
Rat	4	DR	30	d	NR	NR	JV	M	GRO	BDWT	WO	285	—
Mouse	5	DR	10	w	NR	NR	JV	M	GRO	BDWT	WO	362	—
Rat	2	DR	30	d	52	d	JV	M	GRO	BDWT	WO	364	—
Rat	4	GV	14	d	14	d	JV	NR	GRO	BDWT	WO	400	800
Rat	5	GV	14	d	20	d	JV	NR	GRO	BDWT	WO	400	800
Rat	2	FD	14	mo	0	d	JV	NR	GRO	BDWT	WO	431	—
Rat	2	GV	102	d	30	d	LC	F	GRO	BDWT	WO	441	—
Mouse	4	GV	12	d	6	d	JV	M	GRO	BDWT	WO	534	—
Mouse	7	FD	30	d	NR	NR	LC	F	GRO	BDWT	WO	632	1264
Rat	2	DR	126	d	1	d	GE	F	GRO	BDWT	WO	651	—
Rat	2	DR	20	w	10	w	GE	F	GRO	BDWT	WO	750	—

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Mouse	7	FD	28	d	NR	NR	LC	F	GRO	BDWT	WO	1260	2530
Rat	4	FD	18	d	NR	NR	LC	F	GRO	BDWT	WO	1500	
Rat	2	DR	9	d	21	d	JV	M	GRO	BDWT	WO	—	3.30
Cattle	2	FD	283	d	7	mo	JV	M	GRO	BDWT	WO	—	15.0
Rat	3	DR	92	d	25	d	GE	F	MPH	GMPH	TB	—	28.7
Rat	4	DR	7	d	25	d	GE	F	GRO	BDWT	WO	—	29.0
Rat	2	DR	5	d	26	d	JV	F	GRO	BDWT	WO	—	29.0
Rat	2	DR	26	d	22	d	JV	F	GRO	BDWT	WO	—	29.5
Rat	2	DR	14	d	26	d	JV	F	MPH	Other	TA	—	29.9
Rat	2	DR	10	d	26	d	JV	F	GRO	BDWT	WO	—	30.4
Mouse	2	GV	3	w	NR	NR	JV	M	GRO	BDWT	WO	—	46.4
Dog	2	OR	5	w	<1	yr	JV	NR	GRO	BDWT	WO	—	50.0
Shrew	4	FD	31	d	NR	NR	JV	B	GRO	BDWT	WO	—	61.5
Rat	3	GV	58	d	2	d	JV	B	GRO	BDWT	WO	—	100
Pig	2	FD	13	w	4	w	JV	NR	GRO	BDWT	WO	—	173
Rat	2	GV	29	d	2	d	JV	F	GRO	BDWT	WO	—	200
Rat	2	FD	5	w	NR	NR	MA	NR	GRO	BDWT	WO	—	272
Rat	2	GV	6	d	1	d	JV	B	GRO	BDWT	WO	—	328
Rat	2	DR	30	d	27	d	JV	M	GRO	BDWT	WO	—	354
Rat	2	DR	50	d	24	d	JV	M	GRO	BDWT	WO	—	371
Rat	2	GV	28	d	2	d	JV	M	GRO	BDWT	WO	—	400
Rat	4	GV	14	d	18	d	JV	NR	GRO	BDWT	WO	—	400

Table 8-1.5.4 (cont'd). Mammalian Toxicity Data Used to Develop the Eco-SSL

Mammalian Species	No. of Doses	Route of Exposure	Exposure Duration	Duration Units	Age	Age Units	Lifestage	Sex	Effect Type	Effect Measure	Response Site	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)
Mouse	2	DR	45	d	50-100	d	GE	F	GRO	BDWT	WO	—	404
Rat	4	FD	1	w	NR	NR	LC	F	GRO	BDWT	WO	—	442
Rat	2	DR	6	w	14	w	LC	F	GRO	BDWT	WO	—	638
Mouse	4	DR	10	w	11	w	JV	F	GRO	BDWT	WO	—	748
Rat	2	FD	21	d	NR	NR	LC	F	GRO	BDWT	WO	—	991
Rat	2	GV	18	d	2	d	JV	B	GRO	BDWT	WO	—	1000
Rat	2	FD	2	w	0	d	JV	NR	GRO	BDWT	WO	—	1430
Rat	4	GV	14	d	24	d	JV	NR	GRO	BDWT	WO	—	1600
Rat	2	FD	2	w	60-80	d	JV	M	GRO	BDWT	WO	—	2390
Rat	3	GV	14	d	16	d	JV	NR	GRO	BDWT	WO	—	2400
Rat	2	FD	14	d	60	d	JV	M	GRO	BDWT	WO	—	2650

AD = adult; B = both; BDWT = body weight changes; d = days; DEYO = death of young; DR = drinking water; F = female; FD = food; FERT = fertility; GMPH = general morphology; GRO = growth; GV = gavage; JV = juvenile; LC = lactation; M = male; MA = mature; mo = months; MPH = morphology; NR = not reported; ODVP = offspring development; OR = other oral; PG = prostate gland; PROG = progeny counts or numbers; PRWT = progeny weight; RBPH = reproductive behavior; REP = reproduction; RHIS = reproductive organ histology; RSEM = resorbed embryos; RSUC = reproductive success (general); RT = reproductive tissue; SM = sexually mature; SPCL = sperm cell counts; SPCV = sperm cell viability; TA = tail; TB = tibia; TE = testes; TEDG = testes degeneration; TEWT = testes weight; w = weeks; WO = whole organism; yr = years.

Source: U.S. Environmental Protection Agency (2005a)

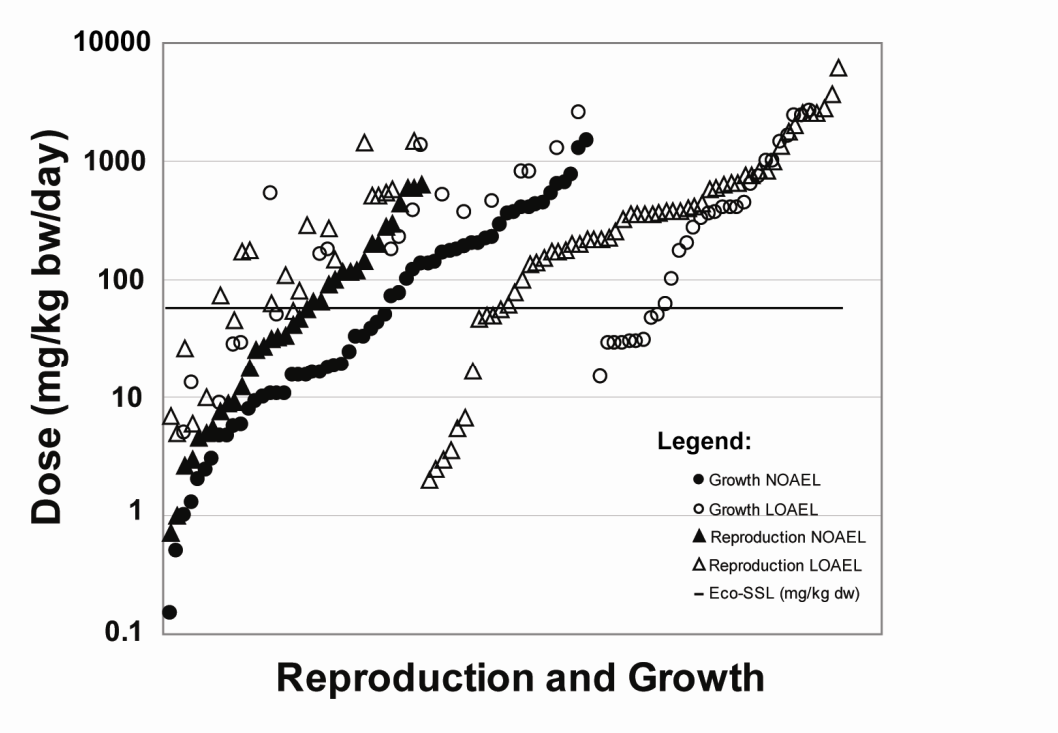

Figure 8-1.5.2. Mammalian toxicity data considered in development of the Eco-SSL.

Source: U.S. Environmental Protection Agency (2005a).

Protection Agency, 2005a) generally do not support this. While five studies reported decreased survival at these levels, 34 other studies reported no mortality or a LOAEL for mortality at significantly higher doses (U.S. Environmental Protection Agency, 2005a). The five studies that supported this low toxic level were conducted on three species (mouse, rat, and cow) and used either gavage or drinking water as the exposure method. The 34 other studies included data on these three species as well as five other species (rabbit, dog, pig, hamster, and shrew) and included gavage and drinking water as well as food ingestion exposure methods. The NOAELs for survival ranged from 3.5 to 3200 mg/kg-day (U.S. Environmental Protection Agency, 2005a). Therefore, the review of data in the Eco-SSL document suggests effects on survival generally would occur at doses greater than those reported to be toxic in the 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a).

8.1.5.5 Recent Studies on the Effects of Lead on Decomposers

Recent studies on effects of Pb to two groups of decomposers are summarized in this subsection. Effects on terrestrial invertebrates, such as earthworms and springtails, are described first, followed by effects on microorganisms.

Effects on Invertebrates

Since the 1986 Pb AQCD, there have been several studies in which terrestrial invertebrates were exposed to Pb in soil. Many of these were reviewed during the development of the Eco-SSLs (U.S. Environmental Protection Agency, 2005a). The relevant information from the Eco-SSL document is described below.

A literature search and review was conducted to identify critical papers published since 2002. Effects on earthworms and other invertebrates observed in studies conducted since the 1986 Pb AQCD are similar to those reported previously: mortality and decreased growth and reproduction (Lock and Janssen, 2002; Davies et al., 2002; Rao et al., 2003; Bongers et al., 2004; Nursita et al., 2005; U.S. Environmental Protection Agency, 2005a).

The literature search completed for terrestrial invertebrate Eco-SSL development identified 179 papers for detailed review, of which 13 met the minimum criteria for further consideration (U.S. Environmental Protection Agency, 2005a). Most of the 18 ecotoxicological endpoints that were further evaluated measured reproduction or survival as the ecologically relevant endpoint. Four of these, representing one species under three different pH test conditions were used to develop the Eco-SSL of 1700 mg/kg soil (Table 8-1.5.5).

In a study designed to test the toxicity of Pb to the earthworm *Eisenia fetida*, Davies et al. (2002) found that the 28-day LC_{50} for Pb in soils contaminated with $Pb(NO_3)_2$ was 4379 ± 356 mg/kg. Twenty-eight day EC_{50} values for weight change and cocoon production were 1408 ± 198 and 971 ± 633 mg/kg, respectively. Significant mortalities were noted at concentrations of 2000 mg/kg. These data are consistent with those reported in the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) for the same species of earthworm.

Nursita et al. (2005) found no mortality and no adverse effects on reproduction (i.e., number of juveniles) of the collembolan *Proisotoma minuta* exposed for 42 days to 300, 750, 1500, or 3000 mg Pb/kg as Pb-nitrate in an acidic (pH = 4.88) sandy loam soil. It was noted that the soils were allowed to equilibrate for 4 weeks after adding the Pb-nitrate before the organisms

Table 8-1.5.5. Invertebrate Toxicity Data Used to Develop the Eco-SSL

Invertebrate Species	Soil pH	% Organic Matter	Toxicity Parameter	Pb in Soil (mg/kg dw)
Collembola *(Folsomia candida)*	6.0	10	MATC[1] (reproduction)	3162
Collembola	4.5	10	MATC (reproduction)	3162
Collembola	5.0	10	MATC (reproduction)	894
Collembola	6.0	10	MATC (reproduction)	894
			Geometric Mean	1682

* MATC = Maximum Acceptable Threshold Concentration, or the geometric mean of the NOEC (no-observed-effect concentration) and LOEC (lowest-observed-effect concentration).

Source: U.S. Environmental Protection Agency (2005a).

were added. The observation of no effect at 3000 mg/kg is consistent with that of Sandifer and Hopkin (1996). Sandifer and Hopkin (1996) determined a NOEC and LOEC for collembolan reproduction of 2000 and 5000 mg/kg, respectively. (A MATC of 3162 mg/kg was used to develop the Eco-SSL).

The remaining 14 toxicity endpoints that were not used to develop the Eco-SSL for invertebrates are presented in Table 8-1.5.6. None of these endpoints were considered eligible for Eco-SSL derivation.

Lock and Janssen (2002) exposed the potworm *Enchytraeus albidus* to Pb, as Pb-nitrate. The 21-day LC_{50} was 4530 mg/kg, and the 42-day EC_{50} for juvenile reproduction was 320 mg/kg. The F1 generation was then grown to maturity in the same concentration soil and subsequently used in a reproduction test. The EC_{50} for the F1 generation (394 mg/kg) was similar to that of the P generation. The authors concluded that the two-generation assay did not increase the sensitivity of the test (Lock and Janssen, 2002). None of the 18 toxicity endpoints evaluated in detail during development of the Eco-SSLs used this species. The LC_{50} reported for the potworm was higher than reported for nematodes and similar to that reported for the earthworm. The EC_{50} for reproduction was lower than reported for the earthworm or collembola.

Table 8-1.5.6. Invertebrate Toxicity Data Not Used to Develop the Eco-SSL

Invertebrate Species	Soil pH	% Organic Matter	Toxicity Parameter	Pb in Soil (mg/kg dw)
Nematode	4	1.14	LC_{50}	285
Nematode	4	1.14	LC_{50}	297
Nematode	4	4.2	LC_{50}	847
Nematode	4	4.2	LC_{50}	1341
Nematode	6.2	1.7	LC_{50}	1554
Nematode	5.1	3.0	LC_{50}	891
Earthworm	6.3	10.0	EC_{50}	1940
Earthworm	6.1	10.0	EC_{50}	1629
Earthworm	6.0	10.0	LC_{50}	3716
Earthworm	6.5	10.0	ILL	1.16
Nematode	4	10	LC_{50}	1434
Nematode	4	10	NOAEC	2235
Nematode	6.1	3.4	LC_{50}	13.9
Nematode	6.2	2.2	LC_{50}	11.6

*NOAEC (no-observed-adverse-effect concentration); LC_{50} (concentration lethal to 50% of test population); EC_{50} (effect concentration for 50% of test population); ILL (incipient lethal level).

Source: U.S. Environmental Protection Agency (2005a).

Recent work by Bongers et al. (2004) cautioned against attributing all toxicity observed in a spiked-soil toxicity test to Pb. They found that the counterion may also contribute to thetoxicity of Pb in the springtail *Folsomia candida*. This may have implications on the interpretation of the Eco-SSL data, because the toxicity of the counterion (nitrate) was not taken into account during Eco-SSL development. Percolation (removal of the counterion) had no statistically significant effect on Pb-chloride toxicity (LC_{50} = 2900 mg/kg for both non-percolated and percolated soil; EC_{50} for reproduction = 1900 mg/kg or 2400 mg/kg for non-percolated or percolated soil, respectively). However, percolation did have a significant effect

on Pb-nitrate toxicity (LC_{50} = 980 mg/kg or 2200 mg/kg for non-percolated and percolated soil, respectively; EC_{50} for reproduction = 580 mg/kg or 1700 mg/kg for non-percolated or percolated soil, respectively). Lead nitrate was more toxic than Pb-chloride for survival and reproduction. However, the toxicity of Pb, from chloride or nitrate, was not significantly different after the counterion was percolated out of the test soil. It is noted that the soil was left for 3 weeks to equilibrate before testing. Lock and Janssen (2002) also found that Pb-nitrate was more toxic than Pb-chloride, and they used Pb-nitrate in their experiments because 1000 mg/kg Pb-chloride did not produce any mortality in their range-finding tests. This difference in chloride and nitrate toxicity has not been found for earthworms (Neuhauser et al., 1985; Bongers et al., 2004).

Rao et al. (2003) exposed the earthworm *Eisenia fetida* to Pb-oxide in an artificial soil with a pH of 6 at the LC_{50} concentration of 11 mg/kg. Exposure for 14 days resulted in a number of effects including body fragmentation, protrusions, rupture of the cuticle, etc. Many of these effects may trigger defensive mechanisms. For example, fragmentation of the affected posterior region was followed by regeneration and a new ectoderm layer was formed to cover affected areas, both of which processes may serve to prevent soil bacteria from further affecting the earthworm (Rao et al., 2003).

Effects on Microorganisms and Microbial Processes

Microorganisms and microbial processes were not included in the Eco-SSL development process (see Attachment 1-2 of OSWER Direction 92857-55 dated November 2003). Many reasons were given, including that it is unlikely that site conditions would only pose unacceptable risk to microbes and not be reflected as unacceptable risks to higher organisms; that the significance of laboratory-derived effects data to the ecosystem is uncertain; and that the spatial (across millimeter distances) and temporal (within minutes to hours) variation makes understanding ecological consequences challenging. Microbial endpoints often vary dramatically based on moisture, temperature, oxygen, and many non-contaminant factors. Therefore, the recommendation arising from the Eco-SSL development process was that risks to microbes or microbial processes not be addressed through the chemical screening process but that they should be addressed within a site-specific risk assessment (U.S. Environmental Protection Agency, 2005a).

Few studies on the effects of Pb to microbial processes have been published since 1986. As the direct toxicity to fungi and bacterial populations are difficult to determine and interpret, indicators for soil communities are often measured as proxies for toxicity (e.g., urease activity in soil). Recent studies of this nature (Doelman and Haanstra, 1986; Wilke, 1989; Haanstra and Doelman, 1991) are summarized in this subsection. The Pb concentrations in these recent studies (1000 to 5000 mg/kg) are consistent with those reported in the 1986 Pb AQCD as associated with effects on microbial processes (750 to 7500 mg/kg).

The effects of Pb-chloride on the processes of nitrification and nitrogen mineralization were studied in a 28-day experiment by Wilke (1989). The authors reported that nitrification was increased by 12 and 16% at levels of 1000 and 4000 mg/kg, respectively, and that nitrogen mineralization was reduced by 32 and 44% at concentrations of 1000 and 4000 mg/kg, respectively.

The effects of Pb on arylsulfatase (Haanstra and Doelman, 1991) and urease activity (Doelman and Haanstra, 1986) in soil were investigated. LC_{50}s for decreases in arylsulfatase activity were reported at Pb concentrations of 3004 and 4538 mg/kg in a silty loam soil, at pH 6 and 8, respectively. The LC_{50} for a decrease in urease activity was 5060 mg Pb/kg in a sandy loam soil.

8.1.5.6 Summary

The current document expands upon and updates knowledge related to the effects of Pb on terrestrial primary producers, consumers, and decomposers.

Primary Producers

The effects of Pb on terrestrial plants include decreased photosynthetic and transpiration rates in addition to decreased growth and yield. The phytotoxicity of Pb is considered to be relatively low, and there are few reports of phytotoxicity from Pb exposure under field conditions. Recently, phytotoxicity data were reviewed for the development of the Eco-SSL (U.S. Environmental Protection Agency, 2005a). Many of the toxicity data presented in the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) are lower than those discussed in the 1986 Pb AQCD, although both documents acknowledged that toxicity is observed over a wide range of concentrations of Pb in soil (tens to thousands of mg/kg soil). This may be due to

many factors, such as the soil conditions (e.g., pH, organic matter) and differences in bioavailability of the Pb in spiked soils perhaps due to lack of equilibration of the Pb solution with the soil after spiking. Most phytotoxicity data continue to be developed for agricultural plant species (i.e., vegetable and grain crops). Few data are available for trees or native herbaceous plants, although two of the five toxicity endpoints used to develop the Eco-SSL were for trees and two were for clover.

Consumers

Effects of Pb on avian and mammalian consumers include decreased survival, reproduction, and growth as well as effects on development and behavior. There remain few field effects data for consumers, except from sites with multiple contaminants, for which it is difficult to attribute toxicity specifically to Pb. Avian and mammalian toxicity data recently were reviewed for the development of Eco-SSLs (U.S. Environmental Protection Agency, 2005a). Many of the toxicity data presented in the Eco-SSL document (U.S. Environmental Protection Agency, 2005a) are lower than those discussed in the 1986 Pb AQCD, although the EPA (U.S. Environmental Protection Agency, 2005a) recognizes that toxicity is observed over a wide range of doses (<1 to >1000 mg Pb/kg bw-day). Most toxicity data for birds have been derived from chicken and quail studies, and most data for mammals have been derived from laboratory rat and mouse studies. Data derived for other species would contribute to the understanding of Pb toxicity, particularly for wildlife species with different gut physiologies. In addition, data derived using environmentally-realistic exposures, such as from Pb-contaminated soil and food, may be recommended. Finally, data derived from inhalation exposures, which evaluate endpoints such as survival, growth, and reproduction, would contribute to understanding the implications of airborne releases of Pb.

Decomposers

Effects of Pb on soil invertebrates include decreased survival, growth, and reproduction. Effects on microorganisms include changes in nitrogen mineralization and enzyme activities. Recent data on Pb toxicity to soil invertebrates and microorganisms are consistent with those reported in the 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a), with toxicity generally observed at concentrations of hundreds to thousands of mg/kg soil. Studies on

microbial processes may be influenced significantly by soil parameters, and the significance of the test results is not clear.

Ecological Soil Screening Levels (Eco-SSLs)

Eco-SSLs are concentrations of contaminants in soils that are protective of ecological receptors (U.S. Environmental Protection Agency, 2005b). They were developed following rigorous scientific protocols and were subjected to two rounds of peer review. The Eco-SSLs for terrestrial plants, birds, mammals, and soil invertebrates are 120, 11, 56, and 1700 mg Pb/kg soil, respectively.

8.1.6 Effects of Lead on Natural Terrestrial Ecosystems

8.1.6.1 Introduction

The concept that organisms are part of larger systems that include both biotic and abiotic components of the environment dates back to the naturalists of the Victorian era. However, the breakthrough in what we now consider the ecosystem approach to ecology occurred in the 1950s and 1960s when E.P. and H.T. Odum pioneered the quantitative analysis of ecosystems (Odum, 1971). This approach encouraged the calculation of energy flows into, out of, and within explicitly defined ecosystems. The rapid development of computer technology aided in the growth of ecosystem ecology by allowing the development and use of increasingly complex models for estimating fluxes that could not be directly measured.

It was not long before the quantitative analysis of ecosystems was extended to examine the flows of nutrients and other chemical compounds. In temperate terrestrial systems, the watershed was identified as a convenient and informative experimental unit (Bormann and Likens, 1967). A major conceptual breakthrough in the watershed approach was that drainage water chemistry could be used as an indicator of the "health" of the ecosystem. In a system limited by nitrogen, for example, elevated concentrations of NO_3^- in drainage waters indicate that the ecosystem is no longer making optimal use of available nutrients.

The ecosystem approach can also be used effectively in the study of trace element biogeochemistry. Input-output budgets can be used to determine whether an ecosystem is a net source or sink of a trace element. Changes to the input-output balance over time can be used to assess the effects of natural or experimental changes in deposition, land use, climate, or other

factors. In addition, examination of fluxes within the ecosystem (in plant uptake, soil solutions, etc.) can be used to understand the processes that are most influential in determining the fate and transport of the trace element.

Many published ecosystem studies include data for 1 to 3 years, the typical duration of research grant funding or doctoral dissertation research. While these studies enrich our understanding of terrestrial ecosystems, the most valuable studies are those that are maintained over many years. Natural variations in climate, pests, animal migrations, and other factors can make inferences from short-term studies misleading (Likens, 1989). To nurture long-term research, the National Science Foundation supports a network of Long-Term Ecological Research (LTER) sites that represent various biomes.

This section describes terrestrial ecosystem research on Pb, focusing on work done since the 1986 Pb AQCD and highlighting key long-term studies. Unfortunately, there are few studies that feature long-term data on trace metal behavior. Therefore, this examination of the effects of Pb on terrestrial ecosystems combines insights from long- and short-term investigations as well as including observations from whole-ecosystem and single-component studies.

8.1.6.2 Effects of Terrestrial Ecosystem Stresses on Lead Cycling

Terrestrial ecosystems may respond to stressors in a variety of ways, including reductions in the vigor and/or growth of vegetation, reductions in biodiversity, and effects on microbial processes. Each of these effects may lead to the "leakage" of nutrients, especially nitrogen, in drainage waters. The reduced vigor or growth of vegetation results in a lower uptake of nitrogen and other nutrients from soils. Reduced biodiversity accompanied by lower total net primary productivity for the ecosystem would also result in a lower nutrient uptake. Effects of stress in microbial populations are less obvious. If the stress reduces microbial activity rates, then nutrients bound in soil organic matter (e.g., organic nitrogen compounds) will likely be mineralized at a lower rate and retained in the system. On the other hand, disturbances such as clear-cutting, ice-storm damage, and soil freezing can result in substantial nutrient losses from soils (Bormann et al., 1968; Likens et al., 1969; Mitchell et al., 1996; Groffman et al., 2001; Houlton et al., 2003).

Since the movement and fate of Pb in terrestrial ecosystems is strongly related to the organic matter cycle (Section 8.1.3), stressors that could lead to disruption or alteration of the

soil organic matter pool are of particular concern in assessing effects of ecosystem stress on Pb cycling. By binding soluble Pb, soil organic matter acts as a barrier to the release of Pb to drainage waters (Wang et al., 1995; Kaste et al., 2003; Watmough and Hutchinson, 2004). As a result, concentrations of Pb in soil solutions and drainage waters tend to be low (Driscoll et al., 1988; Wang et al., 1995; Bacon and Bain, 1995; Johnson et al., 1995b). Through decomposition and leaching, soluble organic matter is released to solution, and with it, some Pb is also mobilized. Wang and Benoit (1996) found that essentially all of the Pb in soil solutions in a hardwood forest in New Hampshire was bound to dissolved organic matter (DOM). This release of soluble Pb does not typically result in elevated surface water Pb concentrations, because (1) organic matter has a relatively long residence time in most temperate soils (Gosz et al., 1976; Schlesinger, 1997), so only a small fraction of the organic matter pool is dissolved at any time; (2) DOM-Pb complexes solubilized in upper soil horizons may be precipitated or adsorbed lower in the soil profile; (3) the DOM to which Pb is bound may be utilized by microbes, allowing the associated Pb to bind anew to soil organic matter. Together, these factors tend to moderate the release of Pb to surface waters in temperate terrestrial ecosystems. However, stressors or disturbances that result in increased release of DOM from soils could result in the unanticipated release of Pb to groundwater and/or surface waters.

8.1.6.2.1 Acidification

The effect of acidification on ecosystem cycling of Pb is difficult to predict. Like most metals, the solubility of Pb is increased at lower pH (Stumm and Morgan, 1995), suggesting that enhanced mobility of Pb should be found in ecosystems under acidification stress. However, reductions in pH may also decrease the solubility of DOM, via protonation of carboxylic functional groups (Tipping and Woof, 1990). Because of the importance of complexation with organic matter to Pb mobility in soils, lower DOM concentrations resulting from acidification may offset the increased solubility of the metal.

In a study of grassland and forest soils at the Rothamsted Experiment Station in England, long-term (i.e., >100 years) soil acidification significantly increased the mobility of Pb in the soil (Blake and Goulding, 2002). However, the increased mobility was only observed in very acid soils, those with pH of <4.5. The fraction of exchangeable Pb (extracted with 0.1 M $CaCl_2$) increased from about 3% to 15% of the total Pb in the most acidified soils. Similarly, the

fraction of organically bound Pb increased from about 2% of total Pb in neutral soils to 12% of total Pb in the most acidified soils. Similarly, Nouri and Reddy (1995) observed higher levels of DTPA-extractable Pb in soils in a loblolly pine forest treated with simulated acid rain, but only in the most acidic treatment, with simulated rain with a pH of 3.5.

Although acidification may increase the mobility of Pb in soils, it is not clear that this Pb is actually moving through or out of the soil profile. In an examination of running waters in Sweden, Johansson et al. (1995) found no relationship between acidification and Pb concentrations and concluded that Pb concentrations were governed by the DOM concentration, which masked any association with acidification. In an in situ lysimeter study, Bergkvist (1986) measured lower concentrations of Pb in soil solutions draining experimentally acidified plots than in unacidified plots. In a laboratory study using large soil columns, Merino and García-Rodeja (1997) observed no effect of experimental acidification on the release of Pb to soil solution. Thus, while acidification may increase the potential mobility of Pb in soils, as indicated by increases in labile soil fractions such as exchangeable and DTPA-extractable Pb, the actual movement of Pb in the soil is limited by DOM solubilization and transport. It is worth noting that in all of these studies, significant effects of acidification were observed for other trace metals (Bergkvist, 1986; Johansson et al., 1995; Merino and García-Rodeja, 1997).

Acidification may enhance Pb export to drainage water in very sandy soils, soils with limited ability to retain organic matter. Studies in the McDonald's Branch watershed in the New Jersey pine barrens, where soil texture is similar to beach sands, suggested little Pb retention in the mineral soil (Swanson and Johnson, 1980; Turner et al., 1985). If acidification results in the mobilization of Pb and organic matter into these mineral soils, then increased streamwater Pb concentrations would likely follow.

8.1.6.2.2 Forest Harvesting

Forest harvesting represents a severe disruption of the organic matter cycle in forest ecosystems. Litter inputs are severely reduced for several years after cutting (e.g., Hughes and Fahey, 1994). The removal of the forest canopy results in reduced interception of precipitation, and, therefore, increased water flux to the soil surface. Also, until a new canopy closes, the soil surface is exposed to increased solar radiation and higher temperatures. Together, the higher moisture and temperature in surface soils tend to increase the rate of organic matter

decomposition. Several studies have estimated decreases of up to 40% in the organic matter content of forest floor soils after clear-cutting (Covington, 1981; Federer, 1984; Johnson et al., 1995a). This loss of organic matter from the forest floor could result in the mobilization of organically complexed Pb. However, observations from clear-cut sites in the United States and Europe indicate that forest harvesting causes little or no mobilization of Pb from forest soils.

At the Hubbard Brook Experimental Forest in New Hampshire, whole-tree harvesting, the most intensive form of clear-cutting, resulted in very small increases in Pb concentrations in soil solutions draining the Oa soil horizon despite substantial reductions in the organic matter mass of that horizon (Fuller et al., 1988; Johnson et al., 1995a). These increases were associated with similarly small increases in dissolved organic carbon (DOC) concentrations in the Oa horizon soil water. Output of Pb from the watershed stream was unaffected by clear-cutting. Similarly, Berthelsen and Steinnes (1995) observed small decreases in the Pb content of the Oa horizon ("H" in the European system of soil classification) in clear-cut sites in Norway, compared to uncut reference sites. This mobilization of Pb from the Oa horizon was accompanied by an increase in the Pb content of the upper mineral soil horizons. The Pb decline in the Oa horizon was accompanied by a decrease in the organic matter content, leading the authors to attribute the Pb dynamics to leaching with DOM. In a study conducted in Wales, Durand et al. (1994) observed lower Pb outputs from a stream draining a clear-cut watershed than from where the stream drained the upper reaches of the watershed, which were uncut. The DOC and H^+ outputs were also lower in the clear-cut area. These patterns persisted in all 5 years of the study.

Forest harvesting is a severe form of ecosystem disturbance, and, thus, it is somewhat surprising that studies of clear-cutting have shown little or no effect on Pb mobility or loss from forest ecosystems. Perhaps the strong complexation behavior of Pb with natural organic matter results in the retention of Pb in forest soils. Even in cases where Pb is mobilized in forest floor soils (Fuller et al., 1988; Berthelsen and Steinnes, 1995), there is no evidence of loss of Pb from the ecosystem, indicating that mineral soils are efficient in capturing and retaining any Pb that is mobilized in the forest floor. Therefore, the principal risk associated with forest harvesting is the loss of Pb in particulate form to drainage waters through erosion.

8.1.6.2.3 Land Use and Industry

Changes in land use also represent potentially significant changes in the cycling of organic matter in terrestrial ecosystems. Conversion of pasture and croplands to woodlands changes the nature and quantity of organic matter inputs to the soil. In temperate climates, forest ecosystems tend to accumulate organic matter in an O horizon on the forest floor, whereas organic matter in grasslands and agricultural fields is concentrated in an A horizon at the soil surface. Andersen et al. (2002) compared the trace metal concentrations in arable fields in Denmark to nearby sites that had been converted to forest land. After 34 years of afforestation, the soils showed no significant difference in Pb concentration or fractionation, despite significant acidification of the soils. Afforestation had no effect on the soil carbon concentration, suggesting that land use change may have little effect on Pb cycling unless soil carbon pools are affected.

Similarly, the introduction of industrial activity may have consequences for organic matter cycling, and subsequently, Pb mobilization. In a rare long-term study of polluted soils, Egli et al. (1999) studied the changes in trace metal concentrations in forest soils at a site in western Switzerland between 1969 and 1993. The site is 3 to 6 km downwind from an aluminum industrial plant that operated between the 1950s and 1991. In the 24-year period of study, the site experienced significant declines in organic carbon in surface (0 to 5 cm depth) and subsurface (30 to 35 cm) soils. In the 30 to 35 cm layer, the organic carbon concentration declined by more than 75%. Extractable Pb (using an ammonium acetate and EDTA mixture) declined by 35% in the same layer. The authors suggested that the Pb lost from the soil had been organically bound. While this study indicates that loss of soil carbon can induce the mobilization and loss of Pb from terrestrial ecosystems, it is also worth noting that the decline in soil Pb was considerably smaller than the decline in organic carbon. This suggests that Pb mobilized during organic matter decomposition can resorb to remaining organic matter or perhaps to alternate binding sites (e.g., Fe and Mn oxides).

The effects of industries that emit Pb to the atmosphere are discussed in Sections 8.1.6.3 and 8.1.6.4 below.

8.1.6.2.4 Climate Change

Atmospheric Pb is not likely to contribute significantly to global climate change. Lead compounds have relatively short residence times in the atmosphere, making it unlikely that they will reach the stratosphere. Also, Pb compounds are not known to absorb infrared radiation and, therefore, are unlikely to contribute to stratospheric ozone depletion or global warming.

Climate change does, however, represent a disturbance to terrestrial ecosystems. Unfortunately, the potential linkages between climate-related stress and Pb cycling are poorly understood. As in the previous examples, effects related to alterations in organic matter cycling may influence Pb migration. For example, an increase in temperature leading to increased rates of organic matter decomposition could lead to temporary increases in DOM concentrations and smaller steady-state pools of soil organic matter. Either of these factors could result in increased concentrations of Pb in waters draining terrestrial ecosystems.

Climate change may also affect the fluctuations of temperature and/or precipitation in terrestrial ecosystems. For example, there is some evidence for recent increases in the frequency of soil freezing events in the northeastern United States (Mitchell et al., 1996). Soil freezing occurs when soils have little to no snow cover to insulate them from cold temperatures and results in an increased release of nitrate and DOC from the O horizons of forest soils (Mitchell et al., 1996; Fitzhugh et al., 2001). Increased DOC losses from O horizons subjected to freezing may also increase Pb mobilization.

Increased fluctuations in precipitation may induce more frequent flooding, with potentially significant consequences for Pb contamination of floodplain ecosystems. Soils collected from the floodplain of the Elbe River, in Germany, contained elevated concentrations of Pb and other trace metals (Kruger and Grongroft, 2004). Tissues of plants from floodplain sites did not, however, contain higher Pb concentrations than control sites. More frequent or more severe flooding would likely result in increased inputs of Pb and other metals to floodplain soils.

8.1.6.3 Effects of Lead Exposure on Natural Ecosystem Structure and Function

The effects of Pb exposure on natural ecosystems are confounded by the fact that Pb exposure cannot be decoupled from other factors that may also affect the ecosystem under consideration. Principal among these factors are other trace metals and acidic deposition.

Emissions of Pb from smelting and other industrial activities are accompanied by other trace metals (e.g., Zn, Cu, Cd) and sulfur dioxide (SO_2) that may cause toxic effects independently or in concert with Pb. Reductions in the use of alkyl-Pb additives in gasoline have resulted in significant decreases in Pb deposition to natural ecosystems in the northeastern United States (Johnson et al., 1995b). However, the period in which Pb deposition has declined (ca. 1975 to the present) has also seen significant reductions in the acidity (i.e., increased pH) of precipitation in the region (Likens et al., 1996; Driscoll et al., 1998). Therefore, changes in ecosystem Pb fluxes may be the result of reduced Pb inputs and/or reduced acidity.

Experimental manipulation studies do not suffer from these confounding effects, because Pb can be added in specific amounts, with or without other compounds. Unfortunately, ecosystem-level manipulations involving Pb additions have not been undertaken. Therefore, we must use observations from field studies of Pb behavior in sites exposed to various forms of Pb pollution to assess the effects of Pb on terrestrial ecosystems. This section includes a discussion of effects of Pb in the structure and function of terrestrial ecosystems. Effects on energy flows (food chain effects) and biogeochemical cycling are discussed in Section 8.1.6.4.

8.1.6.3.1 Sites Affected by Nearby Point Sources of Lead

Natural terrestrial ecosystems near smelters, mines, and other industrial plants have exhibited a variety of effects related to ecosystem structure and function. These effects include decreases in species diversity, changes in floral and faunal community composition, and decreasing vigor of terrestrial vegetation.

All of these effects were observed in ecosystems surrounding the Anaconda smelter in southwestern Montana, which operated between 1884 and 1980 (Galbraith et al., 1995). Soils in affected areas around the Ananconda smelter were enriched in Pb, arsenic, copper, cadmium, and zinc; had very low pH; and were determined to be phytotoxic to native vegetation (Kapustka et al., 1995). The elevated soil arsenic and metal concentrations occurred despite significantly lower organic matter concentrations in affected soils relative to reference sites (Galbraith et al., 1995). Line-transect measurements indicated that affected sites had an average of 6.9 species per 10-m of transect, compared to 20.3 species per 10-m in the reference areas. More than 60% of the reference sites supported coniferous (58%) or deciduous (3%) forest communities, whereas less than 1% of the affected sites retained functioning forest stands. Abundant dead timber and

stumps confirmed that the affected sites were once as forested as the reference sites. Affected grassland sites were also less diverse and had higher abundances of invasive species than reference grasslands. More than 50% of the affected sites were classified as bare ground. The occurrence of bare ground was significantly correlated with the phytotoxicity scores derived by Kapustka et al. (1995), indicating a link between phytotoxicity and the loss of vegetation in the affected area.

Because of the plant community changes near the Anaconda smelter, the vertical diversity of habitats in the affected ecosystems decreased, with only shrubs and soil remaining as viable habitats. Galbraith et al. (1995) also used the Bureau of Land Management's habitat evaluation procedure (HEP) to estimate habitat suitability indices (HSI) for two indicator species, marten (*Martes americana*) and elk (*Cervus elaphus*). The HSI value ranges from 0 (poor habitat) to 1 (ideal habitat). In sites affected by the Anaconda smelter, HSI values for marten averaged 0.0, compared to 0.5 to 0.8 for the reference sites. For elk, affected sites had an average HSI of 0.10, compared to 0.31 at reference sites.

Similar observations were made in the area surrounding Palmerton, Pennsylvania, where two zinc smelters operated between 1898 and 1980. Soils in the area were enriched in Cd, Zn, Pb, and Cu, with concentrations decreasing with distance from the smelter sites (Beyer et al., 1985; Storm et al., 1994). Smelting was determined to be the principal source of Pb in soils in residential and undeveloped areas around Palmerton (Ketterer et al., 2001), which lies on the north side of a gap in Blue Mountain, a ridge running roughly east-west in east-central Pennsylvania. Much of the north-facing side of Blue Mountain within 3 km of the town is bare ground or sparsely vegetated, whereas the surrounding natural landscape is predominantly oak forest (Sopper, 1989; Storm et al., 1994). Biodiversity in affected areas is considerably lower than at reference sites, a pattern attributed to emissions from the smelters (Beyer et al., 1985; Sopper, 1989). The history is complicated, however, by the land use history of the area. Logging and fire in the early 20th century may also have played a role in the changes in the terrestrial ecosystems (Jordan, 1975). Extensive logging occurred after the smelters began operation, suggesting that some of the logging may have been salvage logging in affected areas. Regardless, the smelter emissions appear to have inhibited the regrowth of ecosystems compared to those in nearby unaffected areas. As in Anaconda, MT, the changes in the structure and function of the Palmerton ecosystem changed its suitability as a habitat for fauna that would

normally inhabit the area. Storm et al. (1994) did not find amphibians or common invertebrates in two study sites nearest to the smelters. In the larger study area, they documented elevated concentrations of Pb, Cd, Cu, and Zn in tissues of species ranging in size from red-backed salamanders (*Pletheron cenereus*) to white-tailed deer (*Odocoilius virginianus*).

Metal pollution around a Pb-Zn smelter near Bristol, England has not resulted in the loss of oak woodlands within 3 km of the smelter, despite significant accumulation of Pb, Cd, Cu, and Zn in soils and vegetation (Martin and Bullock, 1994). However, the high metal concentrations have favored the growth of metal-tolerant species in the woodland.

The effects of Pb on terrestrial ecosystems near smelters and other industrial sites decrease downwind from the Pb source. Several studies using the soil Pb burden as an indicator have shown that much of the contamination occurs within a radius of 20 to 50 km around the emission source (Miller and McFee, 1983; Martin and Bullock, 1994; Galbraith et al., 1995; Spurgeon and Hopkin, 1996a; see also Section 8.1.3.). For example, the concentration of Pb in forest litter declined downwind from a Pb-Zn smelter near Bristol, UK, from 2330 to 3050 ppm in a stand 2.9 km from the smelter to 45 to 110 ppm in a stand 23 km from the smelter (Martin and Bullock, 1994). Thus, while sites near point sources of Pb may experience profound effects on ecosystem structure and function, the extent of those effects is limited spatially. Most terrestrial ecosystems are far enough from point sources that long-range Pb transport is the primary mechanism for Pb inputs.

8.1.6.3.2 Sites Affected by Long-Range Lead Transport

Because the effects of anthropogenic Pb emissions tend to be restricted in geographic extent, most natural terrestrial ecosystems in the U.S. sites have Pb burdens derived primarily from long-range atmospheric transport. Pollutant Pb represents a large fraction of the Pb in many of these ecosystems. In particular, many of these sites have accumulated large amounts of Pb in soils. For example, at the Hubbard Brook Experimental Forest in New Hampshire, the amount of Pb in the forest floor was estimated to have increased from about 1.35 kg ha^{-1} in 1926 (before the introduction of alkyl-Pb additives in gasoline) to 10.5 kg ha^{-1} in 1977 (Johnson et al., 1995b). They also estimated the atmospheric Pb deposition from 1926 to 1987 to be 8.7 kg ha^{-1}, an amount that could account for nearly all of the increase in Pb in the forest floor during the period. The input of precipitation Pb to the Hubbard Brook ecosystem in the six decades

spanning 1926 to 1987 was more than half of the total Pb estimated to have been released by mineral weathering in the entire 12,000- to 14,000-year post-glacial period (14.1 kg ha^{-1}: (Johnson et al., 2004)). Other studies employing Pb budgets (Miller and Friedland, 1994; Watmough et al., 2004), and Pb isotopes (Bacon et al., 1995, 1996; Watmough et al., 1998; Bindler et al., 1999; Hansmann and Köppel, 2000; Kaste et al., 2003), have also shown that pollutant Pb, primarily from gasoline combustion, represents a quantitatively significant fraction of labile Pb in temperate soils, especially in the upper, organic-rich horizons.

Despite years of elevated atmospheric Pb inputs and elevated concentrations in soils, there is little evidence that sites affected primarily by long-range Pb transport have experienced significant effects on ecosystem structure or function. Low concentrations of Pb in soil solutions, the result of strong complexation of Pb by soil organic matter, may explain why few ecological effects have been observed. At Hubbard Brook, for example, the concentration of Pb in soil solutions draining the Oa horizon is <0.1 μM and is even lower in solutions draining mineral-soil horizons (Driscoll et al., 1988; Wang et al., 1995). Friedland and Johnson (1985) measured similar concentrations in soil solutions collected from deciduous and spruce-fir stands on Camel's Hump Mountain in Vermont.

In ecosystems where Pb concentrations in soil solutions are low, toxicity levels for vegetation are not likely to be reached regardless of the soil Pb concentration. Furthermore, mycorrhizal infection of tree roots appears to reduce the translocation of Pb from roots to shoots (Marschner et al., 1996; Jentschke et al., 1998). In a study of mycorrhizal and non-mycorrhizal Norway spruce (*Picea abies* (L.) Karst.), mycorrhizal infection of roots was not affected by Pb dose. Some, but not all, species of mycorrhizae showed reductions in the amount of extrametrical mycelium with Pb exposure but only at solution concentrations of 5 μM, a level at least 50 times greater than typical concentrations in forest soils. In a related study, the growth rate of mycorrhizal fungi was unaffected at solution Pb concentrations of 1 and 10 μM, but decreased at 500 μM (Marschner et al., 1999).

Low soil solution Pb concentrations and the influence of mycorrhizal symbionts also result in low uptake of Pb by terrestrial vegetation. The net flux of Pb into vegetation in the northern hardwood forest at Hubbard Brook in the 1980s was estimated as only 1 g ha^{-1} $year^{-1}$ (Johnson et al., 1995b), representing 3% of the precipitation input. Klaminder et al. (2005) also measured a Pb uptake of 1 g ha^{-1} $year^{-1}$ in a spruce-pine forest in northern Sweden. Despite

plant uptake fluxes being very low, they are sensitive to differences and changes in Pb deposition. Berthelsen et al. (1995) observed decreases in the Pb content of stem, twig, leaf, and needle tissues of a variety of tree species in Norway between 1982 and 1992, when atmospheric Pb deposition declined by approximately 70%. They also observed significantly lower Pb concentrations in tree tissues collected in northern Norway versus southern Norway, where atmospheric Pb deposition is greater.

Even at subtoxic concentrations, Pb and other metals may influence species diversity in terrestrial ecosystems. However, little work has been done on the effect of low-level metal concentrations on species diversity. In one study, plant species diversity was positively correlated to the concentration of available Pb in natural and artificial urban meadows in Britain (McCrea et al., 2004). The authors hypothesized that Pb may inhibit phosphorous uptake by dominant species, allowing less abundant (but more Pb-tolerant) ones to succeed.

8.1.6.4 Effects of Lead on Energy Flows and Biogeochemical Cycling

In terrestrial ecosystems, energy flow is closely linked to the carbon cycle. The principal input of energy to terrestrial ecosystems is through photosynthesis, in which CO_2 is converted to biomass carbon. Because of this link between photosynthesis and energy flow, any effect that Pb has on the structure and function of terrestrial ecosystems (as discussed in Section 8.1.6.3.) influences the flow of energy into the ecosystem. This section focuses on how Pb influences energy transfer within terrestrial ecosystems, which begin with the decomposition of litter and other detrital material by soil bacteria and fungi, and cascade through the various components of the detrital food web. Because the mobility of Pb in soils is closely tied to organic matter cycling, decomposition processes are central to the biogeochemical cycle of Pb. This section concludes with a discussion of how biogeochemical cycling of Pb has changed in response to the changing Pb inputs to terrestrial ecosystems.

8.1.6.4.1 Effects of Lead on Detrital Energy Flows

Lead can have a significant effect on energy flows in terrestrial ecosystems. At some sites severely affected by metal pollution, death of vegetation can occur, dramatically reducing the input of carbon to the ecosystem (Jordan, 1975; Galbraith et al., 1995). Subsequently, wind and

erosion may remove litter and humus, leaving bare mineral soil, a nearly sterile environment in which very little energy transfer can take place (Little and Martin, 1972; Galbraith et al., 1995).

At Pb-affected sites that can retain a functioning forest stand, the rate of decomposition of litter may be reduced, resulting in greater accumulation of litter on the forest floor than in unpolluted stands. Numerous investigators have documented significant declines in litter decomposition rates (Cotrufo et al., 1995; Johnson and Hale, 2004) and/or the rate of carbon respiration (Laskowski et al., 1994; Cotrufo et al., 1995; Saviozzi et al., 1997; Niklínska et al., 1998; Palmborg et al., 1998; Aka and Darici, 2004) in acid- and metal-contaminated soils or soils treated with Pb. The resulting accumulation of organic matter on the soil surface can be dramatic. For example, an oak woodland 3 km from a smelter in Bristol, England had a litter layer mass 10 times greater than the mass in a similar stand 23 km from the smelter (Martin and Bullock, 1994).

Lower decomposition rates in polluted ecosystems are the result of the inhibition of soil bacteria and fungi and its effects on microbial community structure (Bååth, 1989). Kuperman and Carreiro (1997) observed 60% lower substrate-induced respiration in heavily polluted grassland soils near the U.S. Army's Aberdeen Proving Ground in Maryland. This decline in carbon respiration was associated with 81% lower bacterial biomass and 93% lower fungal biomass. Similar declines in the activities of carbon-, nitrogen-, and phosphorus-acquiring enzymes were also observed. Such dramatic effects have only been observed in highly contaminated ecosystems. In a less contaminated grassland site near a Pb factory in Germany, Chander et al. (2001) observed a lower ratio of microbial biomass carbon to soil organic carbon in polluted soils. The ratio of basal respiration to microbial biomass (the "metabolic quotient," qCO_2) declined with increasing metal concentration, though this observation depended on the procedure for measuring microbial biomass (substrate-induced respiration versus fumigation-extraction). The combined effect of lower microbial biomass per unit soil carbon and similar or lower qCO_2 on polluted sites indicates that the ability of soil microorganisms to process carbon inputs is compromised by metal pollution.

The type of ecosystem also plays a role in determining the effects of Pb and other metals on the microbial processing of litter. Forest soils in temperate zones accumulate organic matter at the soil surface to a greater degree than in grasslands. This organic-rich O horizon can support a large microbial biomass; but it is also an effective trap for Pb inputs, because of the association

between Pb and soil organic matter. At highly contaminated forest sites, microbial biomass and enzyme activities may be depressed (Fritze et al., 1989; Bååth et al., 1991), causing slower decomposition of the litter.

In addition to effects on decomposition and carbon transformations, Pb and other trace metals can also influence key nitrogen cycling processes. Studies in the 1970s demonstrated that Pb and other metals inhibit the mineralization of nitrogen from soil organic matter and nitrification (Liang and Tabatabai, 1977, 1978), resulting in lower nitrogen availability to plants. More recent research has documented significant inhibitory effects of Pb and other metals on the activities of several enzymes believed to be crucial to nitrogen mineralization in soils (Senwo and Tabatabai, 1999; Acosta-Martinez and Tabatabai, 2000; Ekenler and Tabatabai, 2002). This suggests that the inhibitory effect of Pb and other metals is broad-based, and not specific to any particular metabolic pathway. In reducing environments, the rate of denitrification is also depressed by trace metals. Fu and Tabatabai (1989) found that 2.5 μmol g^{-1} of Pb (ca. 500 mg/kg^{-1}) was sufficient to cause 0, 27, and 52% decreases in nitrogen reductase activity in three different soils.

Metal pollution can also affect soil invertebrate populations. Martin and Bullock (1994) observed lower abundances of a variety of woodlice, millipedes, spiders, insects, and earthworms in an oak woodland site 3 km from a Pb-Zn smelter in Bristol, England, compared to a reference site 23 km from the smelter. The differences were most dramatic when expressed per unit mass of litter. Several species that were abundant in the reference site were not found in the contaminated woodland. For example, the abundance of the woodlice *Trichoniscus pusillus* was 151 individuals per m^2 in the reference woodland, but none were found in the contaminated soils. This was also true of 2 of the 3 millipede species, and 4 of the 5 earthworm species studied. At six sites within 1 km from the smelters, no earthworms were present at all (Spurgeon and Hopkin, 1996a). Contamination at this site has apparently reduced both the population and biodiversity of the soil invertebrate community.

The effect of metal pollution on soil invertebrates may be a threshold-type response. In a study conducted in woodlands near two zinc smelters in Noyelles-Godault, in northern France, soils at the most polluted site were devoid of mites and millipedes, while the remaining sites had diversity measures similar to control sites (Grelle et al., 2000).

While Pb pollution affects the population and diversity of soil fauna, there is little evidence of significant bioaccumulation of Pb in the soil food web (see also Section 8.1.4.). In the Bristol, England study, Pb concentrations in earthworms were lower than soil Pb concentrations and much lower than litter Pb concentrations (Martin and Bullock, 1994). Litter-dwelling mites had Pb concentrations that were 10% of the average litter concentration. The predator centipedes *Lithobius forficatus* and *L. variegatus* had mean Pb concentrations of 18.6 and 44.0 mg kg^{-1}, respectively, two orders of magnitude lower than the Pb concentration of litter (2193 mg kg^{-1}) and lower than the concentrations of their known prey species. In a study conducted in a Norway spruce forest affected primarily by automobile exhaust from a nearby highway, earthworms had Pb concentrations similar to the soil (Roth, 1993). Almost all of the litter decomposers, however, had Pb concentrations that were less than 20% of the litter. All but 3 of the zoophagous arthropods had Pb concentrations that were less than 40% of their prey; the remaining 3 had Pb concentrations similar to their prey. Because of the absence of significant bioaccumulation in the soil food web, predator species will be affected by Pb pollution primarily through effects on the abundance of their prey (Spurgeon and Hopkin, 1996b).

Taken as a whole, ecosystem-level studies of the soil food web indicate that Pb can affect energy flows in terrestrial ecosystems through two principal mechanisms. In the most severely polluted sites, the death of primary producers directly decreases the flow of energy into the ecosystems. More commonly, the accumulation of toxic levels of Pb or other metals in litter and soil decreases the rate of litter decomposition through decreases in microbial biomass and/or respiration. These reductions can subsequently affect higher trophic levels that depend on these organisms. It is important to note that sites that have exhibited significant disruption to energy flows and the terrestrial food web are sites that have experienced severe metal contamination and adverse effects from SO_2 from smelters or other metals-related activities.

Lead Dynamics in Terrestrial Ecosystems

Lead inputs to terrestrial ecosystems in the United States have declined dramatically in the past 30 years, primarily because of the almost complete elimination of alkyl-Pb additives in gasoline in North America. Also, Pb emissions from smelters have declined as older plants have been shut down or fitted with improved emissions controls. Unfortunately, there are few long-term data sets of precipitation inputs to terrestrial ecosystems. At the Hubbard Brook

Experimental Forest, in New Hampshire, Pb input in bulk deposition declined by more than 97% between 1976 and 1989 (Johnson et al., 1995b). Studies of freshwater sediments also indicate a dramatic decline in Pb inputs since the mid-1970s (Graney et al., 1995; Johnson et al., 1995b; Farmer et al., 1997; Brännvall et al., 2001a,b).

Reported concentrations of Pb in waters draining natural terrestrial ecosystems have always been low (Wang et al., 1995; Bacon and Bain, 1995; Johnson et al., 1995b; Vinogradoff et al., 2005), generally less than 1 ng L^{-1}, even at moderately polluted sites (Laskowski et al., 1995). Consequently, most terrestrial ecosystems in North America and Europe remain sinks for Pb despite reductions in atmospheric Pb deposition of more than 95%. At Hubbard Brook, for example, the input of Pb in bulk precipitation declined from 325 g ha^{-1} $year^{-1}$ between 1975 and 1977 compared to 29 g ha^{-1} $year^{-1}$ between 1985 and 1987 (Johnson et al., 1995b). During the same period, the output of Pb in stream water declined from 6 g ha^{-1} $year^{-1}$ to 4 g ha^{-1} $year^{-1}$. Thus, despite the decline in Pb input, 85% of the incoming Pb was still retained in the terrestrial ecosystem in the later time period. Similar observations have been made in Europe, where the use of leaded gasoline has also declined in the last few decades. At the Glensaugh Research Station in Scotland, the input of Pb to the forest ecosystem was estimated as 42.6 g ha^{-1} $year^{-1}$ between 2001 and 2003, about six times the stream export of 7.2 g ha^{-1} $year^{-1}$ (Vinogradoff et al., 2005). Similarly, Huang and Matzner (2004) reported a throughfall flux of 16.5 g ha^{-1} $year^{-1}$ at the forested Lehstenbach catchment in Bavaria, about six times the efflux in runoff of 2.82 g ha^{-1} $year^{-1}$.

Lead pollution has resulted in the accumulation of large Pb burdens in terrestrial ecosystems (see Sections 8.1.3. and 8.1.6.3.2). Despite reductions in emissions, this accumulation of Pb continues, though at markedly lower rates. The large pool of Pb bound in soils may potentially be a threat to aquatic ecosystems, depending on its rate of release from the soil. Early estimates of the residence time of Pb in the forest floor ranged from 220 to 5,000 years (Benninger et al., 1975; Friedland and Johnson, 1985; Turner et al., 1985). However, more recent literature suggests that Pb is transported more rapidly within soil profiles than previously believed. The pool of Pb in forest floor soils of the northeastern United States declined significantly in the late 20th century. Friedland et al. (1992) reported a 12% decline in the amount of Pb in forest floor soils at 30 sites in the region between 1980 and 1990, a much greater decline than would be expected for a pool with a residence time of 220 to 5,000 years.

At Hubbard Brook, the pool of Pb in the forest floor declined by 29% between 1977 and 1987, an even more rapid rate of loss than reported by Friedland et al. (1992). More recently, Evans et al. (2005) reported significant declines in the Pb content of forest floor soils in the northeastern United States and eastern Canada between 1979 and 1996. The magnitude of the decrease in Pb content was greatest at their sites in southern Vermont, and smallest at sites on the Gaspe Peninsula in Quebec, reflecting the historic gradient in Pb deposition in the region.

Since drainage water Pb concentrations remain low, the Pb released from forest floor soils in the past has been largely immobilized in mineral soils (Miller and Friedland, 1994; Johnson et al., 1995b; Johnson and Petras, 1998; Watmough and Hutchinson, 2004; Johnson et al., 2004). This is supported by evidence from Pb-isotope analyses. Gasoline-derived Pb has a ^{206}Pb:^{207}Pb ratio that can be easily discriminated from Pb in the rocks from which soils are derived. Using isotopic mixing models with gasoline-Pb and Pb in soil parent materials as end members, a number of researchers have documented the accumulation of pollutant Pb in mineral soils (Bindler et al., 1999; Kaste et al., 2003; Watmough and Hutchinson, 2004; Bacon and Hewitt, 2005; Steinnes and Friedland, 2005). In a hardwood stand on Camel's Hump Mountain in Vermont, as much as 65% of the pollutant Pb deposited to the stand had moved into mineral horizons by 2001 (Kaste et al., 2003). In a spruce-fir stand, containing a thicker organic forest floor layer, penetration of pollutant Pb into the mineral soil was much lower.

This recent research has resulted in a reevaluation of the turnover time of Pb in forest floor soils. The Camel's Hump data suggest that Pb resides in the forest floor of deciduous stands for about 60 years and about 150 years in coniferous stands (Kaste et al., 2003). These values are somewhat greater than those published previously by Miller and Friedland (1994), who used a Pb budget approach. Extremely rapid turnover of Pb was observed in some hardwood forest floor soils in south-central Ontario (Watmough et al., 2004). Their estimated turnover times of 1.8 to 3.1 years are much lower than any other published values, which they attribute to the mull-type forest floor at their sites. Mull-type forest floors are normally underlain by organic-rich A horizons, capable of immobilizing Pb released from the forest floor. Indeed, at the same site in Ontario, Watmough and Hutchinson (2004) found that 90% of the pollutant Pb could be found in this A horizon.

The time period over which the accumulated Pb in soils may be released to drainage waters remains unclear. If Pb moves as a pulse through the soil, there may be a point in the

future at which problematic Pb concentrations occur. However, several authors have argued against this hypothesis (Wang and Benoit, 1997; Kaste et al., 2003; Watmough et al., 2004), contending that the strong linkage between Pb and DOM will result in a temporally dispersed release of Pb in the form of Pb-DOM complexes. Thus, the greatest threat is likely to be in the most highly contaminated areas surrounding point sources of Pb, where the amount of Pb accumulated in the soil is high, and the death of vegetation has resulted in reduced soil organic matter levels.

8.1.6.5 Summary

Atmospheric Pb pollution has resulted in the accumulation of Pb in terrestrial ecosystems throughout the world. In the United States, pollutant Pb represents a significant fraction of the total Pb burden in soils, even in sites remote from smelters and other industrial plants. However, few significant effects of Pb pollution have been observed at sites that are not near point sources of Pb. Evidence from precipitation collection and sediment analyses indicates that atmospheric deposition of Pb has declined dramatically (>95%) at sites unaffected by point sources of Pb, and there is little evidence that Pb accumulated in soils at these sites represents a threat to groundwaters or surface water supplies.

The highest environmental risk for Pb in terrestrial ecosystems exists at sites within about 50 km of smelters and other Pb-emitting industrial sites. Assessing the risks specifically associated with Pb is difficult, because these sites also experience elevated concentrations of other metals and because of effects related to SO_2 emissions. The concentrations of Pb in soils, vegetation, and fauna at these sites can be two to three orders of magnitude higher than in reference areas (see Sections 8.1.3. and 8.1.4.). In the most extreme cases, near smelter sites, the death of vegetation causes a near-complete collapse of the detrital food web, creating a terrestrial ecosystem in which energy and nutrient flows are minimal. More commonly, stress in soil microorganisms and detritivores can cause reductions in the rate of decomposition of detrital organic matter. Although there is little evidence of significant bioaccumulation of Pb in natural terrestrial ecosystems, reductions in microbial and detritivorous populations can affect the success of their predators. Thus, at present, industrial point sources represent the greatest Pb-related threat to the maintenance of sustainable, healthy, diverse, and high-functioning terrestrial ecosystems in the United States.

8.2 AQUATIC ECOSYSTEMS

8.2.1 Introduction

The overall intent of Section 8.2 is to provide sufficient information to support development of an air quality criterion for lead that is protective of aquatic ecosystems. To achieve this objective, the logical starting points are to (1) gain a general understanding of the current distribution and concentrations of lead in the aquatic environment and (2) identify the threshold levels for lead effects on aquatic populations, communities, and ecosystems. For this latter objective, compatible with the EPA's continuing goal of developing environmental criteria using a holistic approach, development of air quality criteria should be integrated with EPA's current ambient water quality criteria (AWQC)[1] for lead and sediment quality benchmarks. Ambient water quality criteria for lead and other chemicals represent surface water concentrations that are intended to be protective of aquatic communities, including recreationally and commercially important species. The EPA derives AWQC to provide guidance to States and Tribes that are authorized to establish water quality standards under the Clean Water Act (CWA). Similarly, EPA has recommended sediment quality benchmarks for lead and other divalent metals, although not truly criteria, that represent concentrations in sediment that are derived to be protective of benthic (sediment) organisms. As summarized further below and in subsequent sections, the EPA has increasingly focused on developing AWQC and sediment quality benchmarks for lead and other metals that account for the bioavailability of the metal to aquatic life. These criteria and benchmark concentrations in water and sediment represent appropriate starting points to ensure that air quality criteria for lead are adequately protective of aquatic life.

Since publication of the 1986 air quality criteria document for lead, knowledge has expanded on the fate and effects of lead in aquatic ecosystems and on the distribution and concentrations of lead in surface waters throughout the United States. In addition, chemical, physical, and biological properties of lead are discussed. The following provides a general overview of the key information from Section 8.2.

[1]Lead AWQC are currently in the process of being updated.

8.2.1.1 Methodologies in Aquatic Ecosystem Research

Ambient Water Quality Criteria and Bioavailability

The primary form of lead in freshwater and marine environments is divalent lead (Pb^{2+}). In surface waters, the bioavailability of lead to aquatic biota is driven by a variety of factors, including calcium, dissolved organic carbon (DOC), pH, alkalinity, and total suspended solids (TSS). Accounting for the influence of calcium and magnesium ions on lead bioavailability, the current AWQC for lead are normalized to the hardness of the receiving water (Table 8-2.1.1). More recently, the biotic ligand model (BLM), which considers the binding of free metal ion to the site of toxic action and competition between metal species and other ions, has been developed to predict the toxicity of several metals under a variety of water quality conditions. The BLM has been incorporated into the draft AWQC for copper and is currently being researched for lead.

Table 8-2.1.1. Summary of Lead Ambient Water Quality Criteria for Freshwater Organisms at Different Hardness Levels

Hardness (mg/L as CaCO3)	Acute Criterion (µg/L)	Chronic Criterion (µg/L)
50	34	1.3
100	82	3.2
200	200	7.7

Sediment Quality Benchmarks and Bioavailability

As in surface waters, there are a number of factors in sediment that can influence lead bioavailability to benthic (sediment) organisms. Although sediment quality criteria have not been formally adopted, the EPA has published an equilibrium partitioning procedure for developing sediment criteria for metals (U.S. Environmental Protection Agency 2005c). Equilibrium partitioning (EqP) theory predicts that metals partition in sediment between acid volatile sulfide, pore water, benthic organisms, and other sediment phases, such as organic carbon. When the sum of the molar concentrations of simultaneously extracted metal (∑SEM)

minus the molar concentration of AVS is less than zero, it can accurately be predicted that sediments are not toxic because of these metals. Further, if ∑SEM-AVS is normalized to the fraction of organic carbon (i.e., (∑SEM-AVS)/f_{OC}), mortality can be more reliably predicted by accounting for both the site-specific organic carbon and AVS concentrations (Table 8-2.1.2). An alternative approach for developing sediment quality guidelines is to use empirical correlations between metal concentrations in bulk sediment to associated biological effects, based on sediment toxicity tests (Table 8-2.1.2). These guidelines are based on total metal concentrations in sediment and do not account for the bioavailability of metals between sediments.

Table 8-2.1.2. Summary of Sediment Quality Benchmarks and Guidelines for Lead

Benchmark/ Guideline Type	Source	Effect Level	Value
Equilibrium partitioning	U.S. Environmental Protection Agency (2005c)	Low risk of adverse biological effects	(SEM-AVS)/f_{OC} < 130 µmol/g_{OC}
		May have adverse biological effects	130 µmol/g_{OC} < (SEM-AVS)/f_{OC} < 3,000 µmol/g_{OC}
		Adverse biological effects expected	(SEM-AVS)/f_{OC} > 3,000 µmol/g_{OC}
Bulk sediment	MacDonald et al. (2000)	TEC	35.8 µg/g dry wt.
		PEC	128 µg/g dry wt.
	Ingersoll et al. (1996)	ERL	55 µg/g dry wt.
		ERM	99 µg/g dry wt.
	Long et al. (1995)	ERL	46.7 µg/g dry wt.
		ERM	218 µg/g dry wt.

AVS = Acid volatile sulfide; ERL = Effects range – low (sediment concentration below which adverse effects are rarely observed or predicted among sensitive species, Long et al. [1995]); ERM = Effects range – median (sediment concentration above which effects are frequently or always observed or predicted among most species, Long et al. [1995]); oc = Organic carbon (f_{OC} = fraction organic carbon, g_{OC} = grams organic carbon); PEC = Probably effect concentration (sediment concentration above which harmful effects are likely to be observed, MacDonald et al. [2000]); SEM = Simultaneously extracted metal; TEC = Threshold effect concentration (sediment concentration below which harmful effects are unlikely to be observed, MacDonald et al. [2000])

8.2.1.2 Distribution of Lead in Aquatic Ecosystems

Speciation of Lead in Aquatic Ecosystems

The speciation of lead in the aquatic environment is controlled by many factors, such as, pH, salinity, sorption, and biotransformation processes. Lead is typically present in acidic aquatic environments as $PbSO_4$, $PbCl_4$, ionic lead, cationic forms of lead hydroxide, and ordinary hydroxide $Pb(OH)_2$. In alkaline, waters common species of lead include anionic forms of lead carbonate $Pb(CO_3)$ and hydroxide $Pb(OH)_2$. In freshwaters, lead typically forms strong complexes with inorganic OH^- and CO_3^{2-} and weak complexes with Cl^- (Bodek et al., 1988; Long & Angino, 1977). The primary form of lead in freshwaters at low pH ($\leq$6.5) is predominantly Pb^{2+} and less abundant inorganic forms include $Pb(HCO)_3$, $Pb(SO_4)_2^{2-}$, PbCl, $PbCO_3$, and $Pb_2(OH)_2CO_3$. At higher pH ($\geq$7.5) lead forms hydroxide complexes ($PbOH^+$, $Pb(OH)_2$, $Pb(OH)_3^-$, $Pb(OH)_4^{2-}$). Lead speciation in seawater is a function of chloride concentration and the primary species are $PbCl_3^- > PbCO_3 > PbCl_2 > PbCl^+$ > and $Pb(OH)^+$ (Fernando, 1995).

Lead sorption to suspended or bed sediments or suspended organic matter typically increases with increasing pH, increasing amounts of iron or manganese; and with the polarity of particulate matter (e.g., clays). Adsorption decreases with water hardness (Syracuse Research Corporation [SRC], 1999). At higher pH, lead precipitates as $Pb(OH)^+$ and $PbHCO_3^+$ into bed sediments (Weber, 1993). Conversely, at low pH, lead is negatively sorbed (repelled from the adsorbent surface) (U.S. Environmental Protection Agency, 1979; Gao et al., 2003). In addition, lead may be remobilized from sediment due to a decrease in metal concentration in the solution phase, complexation with chelating agents (e.g., EDTA), and changing redox conditions (Gao et al., 2003). Changes in water chemistry (e.g., reduced pH or ionic composition) can cause sediment Pb to become re-mobilized and potentially bioavailable to aquatic organisms (Weber, 1993). Methylation may result in lead's remobilization and reintroduction into the aqueous environment compartment and its subsequent release into the atmosphere (SRC, 1999). However, methylation is not a significant environmental pathway controlling lead fate in the aquatic environment.

Lead Concentrations in United States Surface Waters

Nationwide lead data in surface waters, from 1991 onward, were compiled using the United States Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) database. Data were compiled from locations categorized as "ambient" or "natural." Ambient refers to data collected from all sampling locations, while natural referred to data collected from sampling locations categorized as forest, rangeland, or reference. Summary statistics for surface water, sediment (bulk, <63 µm), and fish tissue (whole body and liver) are summarized in Table 8-2.1.3. Overall atmospheric sources of lead are generally decreasing as regulations have removed lead from gasoline and other products (Eisenreich et al., 1986); however, elevated lead concentrations remain near sites with ongoing sources, such as near mining wastes or wastewater effluents.

Table 8-2.1.3. Summary of Lead Concentrations in United States Surface Water, Sediment, and Fish Tissue

	Surface Water – Dissolved (µg/L)		Sediment – Bulk, <63 µM (µg/g dry wt.)		Fish Tissue (µg/g dry wt.)			
					Whole Organism		Liver	
Statistic	**Ambient**	**Natural**	**Ambient**	**Natural**	**Ambient**	**Natural**	**Ambient**	**Natural**
n	3,445	430	1,466	258	332	93	559	83
%ND	86	88	0.48	1.2	39	51	71	89
Min	0.04	0.04	0.50	0.50	0.08	0.08	0.01	0.01
Mean	0.66	0.52	120	109	1.03	0.95	0.36	0.28
95th %ile	1.10	0.50	200	162	1.06	1.26	3.24	2.50
Max	29.78	8.40	12,000	12,000	22.6	22.6	12.7	3.37

%ND = Percentage not detected

In addition to directly measuring lead concentrations in various aquatic compartments, it is useful to study the vertical distribution of lead. Sediment profiling and core dating is a method used to determine the extent of accumulation of atmospheric lead and provides information on potential anthropogenic sources. Sediment concentration profiles are typically coupled with lead isotopic analysis. The isotope fingerprinting method utilizes measurements of the abundance of

common lead isotopes (^{204}Pb, ^{206}Pb, ^{207}Pb, ^{208}Pb) to distinguish between natural lead over geologic time and potential anthropogenic sources. Studies of sediment profiles have suggested that observed increases in lead concentrations in the upper sediment layer are concomitant with increases in anthropogenic inputs (Bloom and Crecelius, 1987; Case et al., 1989; Ritson et al., 1999; Chillrud et al., 2003). Isotopic ratios have been used to link increases in sediment concentrations with specific anthropogenic sources and to estimate historic records of lead fluxes to surface waters and sediments (Flegal et al., 1987, 1989; Blais, 1996; Bindler et al., 1999).

8.2.1.3 Species Response/Mode of Action

Lead Uptake

Lead can bioaccumulate in the tissues of aquatic organisms through ingestion of food and water, and adsorption from water, and can subsequently lead to adverse effects if exposed to sufficiently high concentrations (Vink, 2002). The accumulation of lead is influenced by pH and decreasing pH favors bioavailability and bioaccumulation. In general, aquatic organisms have three bioaccumulation strategies for lead: (1) significant accumulation with a low rate of loss; (2) excretion of lead roughly in balance with the availability of the metal in the environment; and (3) weak net accumulation due to a very low metal uptake rate and no significant excretion (Rainbow 1996). Bioconcentration factors (BCFs) have been reported in the scientific literature for various organisms and range from 840 – 20,000 (aquatic plants), 499 – 3,670 (aquatic invertebrates), and 42 – 45 (fish). Organisms that bioaccumulate lead with little excretion must partition the metal such that it has limited bioavailability, otherwise toxicity will occur if a sufficiently high concentration is reached. As previously mentioned, the biotic ligand model (BLM) has been developed to explore the speciation of metals and to understand how metals accumulate and cause effects in aquatic organisms.

Resistance Mechanisms

Aquatic organisms have various methods to resist the toxic effects of metals such as lead. Resistance processes include detoxification and avoidance responses. Mechanisms of resistance and detoxification vary among aquatic biota. These processes can include translocation, excretion, chelation, adsorption, and vacuolar storage and deposition. For example, protists and plants produce intracellular polypeptides that form complexes with lead (Zenk, 1996; Morelli

and Scarano, 2001). Some macrophytes and wetland plants have developed translocation strategies for tolerance and detoxification (Knowlton et al., 1983; Deng et al., 2004). Various aquatic invertebrates may sequester lead in the exoskeleton (Boisson et al., 2002; Knowlton et al., 1983) or have developed specialized excretion processes (Vogt and Quinitio, 1994). Fish scales and mucous may chelate lead in the water column and potentially reduce lead uptake (Coello and Khan, 1996).

Avoidance responses are actions performed to evade a perceived threat. Some aquatic organisms have been shown to be quite adept at avoiding lead in aquatic systems, while others seem incapable of detecting its presence. Snails have been shown to be sensitive to lead, and avoid it at high concentrations (Lefcort et al., 2004). Conversely, anuran (frog and toad) species lack an avoidance response up to 1000 µg Pb/L (Steele et al., 1991). Fish avoidance of chemical toxicants has been well established, and is a dominant sublethal response in polluted waters (Svecevičius, 2001). However, studies examining avoidance behaviour of lead in fish are lacking. In addition to the presence of toxic metals, light and pH, can also alter preference-avoidance responses.

Physiological Effects of Lead

Physiological effects of lead on aquatic biota can occur at the biochemical, cellular and tissue levels of organization. Lead has been shown to affect brain receptors in fish (Rademacher et al. 2005) and serum enzyme activity (e.g., EROD and ALAD) in fish and amphibians (Kutlu and Susuz, 2004; Blasco and Puppo, 1999; Gill et al., 1991; Vogiatzis and Loumbourdis, 1999). Studies examining the effects of lead on fish blood chemistry have indicated alterations from acute and chronic exposures ranging from 100 to 10,000 µg/L (Gill et al., 1991; Allen, 1993; Gopal et al., 1997). Lead exposure has also been shown to negatively affect the growth of aquatic invertebrates (Arai et al., 2002).

Factors that Modify Organism Response to Lead

There are several factors that may influence organism response to lead exposure. These may include the size or age of an organism, genetics, environmental factors (e.g., pH, salinity), nutrition, and the presence of other contaminants. Lead accumulation in living organisms is controlled, in part, by metabolic rates (Farkas et al., 2003) and by the physiological conditions of

an organism. Relationships between age, size and lead body burden in aquatic invertebrates and fish are variable and depend on many environmental variables (e.g., exposure) (Farkas et al., 2003). For example, examination of lead exposure (up to 100 μg/L) in aquatic invertebrates showed little relationship between body size and lead accumulation (MacLean et al., 1996; Canli and Furness, 1993) while lead accumulation and fish size was found to be positively correlated (Douben, 1989; Köck et al., 1996).

The genetics of an organism and/or population may alter the response to lead exposure through one of two processes: (1) a contaminant may influence selection, by selecting for certain phenotypes that enable populations to better cope with the chemical, or (2) a contaminant can be genotoxic, meaning it can produce alterations in nucleic acids at sublethal exposure concentrations, resulting in changes in hereditary characteristics or DNA inactivation (Shugart, 1995). Genetic selection has been observed in aquatic organisms due to lead tolerance. Because tolerant individuals have a selective advantage over vulnerable individuals in polluted environments, the frequency of tolerance genes will increase in exposed populations over time (Beaty et al., 1998). Several studies have shown that heavy metals can alter population gene pools resulting in decreased genetic diversity (Duan et al., 2000; Kim et al., 2003). Laboratory studies have shown that exposure to lead at 10 mg Pb^{2+}/mL of blood leads to chromosomal aberrations in some aquatic organisms (Cestari et al., 2004). Low level (50 μg/L) lead exposure in water over four weeks resulted in DNA strand breakage in the freshwater mussel *Anodonta grandis* (Black et al., 1996). More recently, Cestari et al. (2004) observed similar results (increase in the frequency of chromosomal aberrations and DNA damage in kidney cell cultures) in fish (*Hoplias malabaricus*) that were fed lead contaminated food over 18, 41 and 64 days.

Environmental factors can alter the availability, uptake and toxicity of lead to aquatic organisms. Van Hattum et al. (1996) studied the influence of abiotic variables, including dissolved organic carbon (DOC) on lead concentrations in freshwater isopods and found that as DOC concentrations increased, BCFs decreased in *P. meridianus* and *A. aquaticus*, indicating that DOC acts to inhibit the availability of lead to these isopods. Schwartz et al. (2004) collected natural organic matter (NOM) from several aquatic sites across Canada and investigated the effects of NOM on lead toxicity in rainbow trout (*Oncorhynchus mykiss*). The results showed that NOM in test water almost always increased LT50 (time to reach 50% mortality), and optically dark NOM tended to decrease lead toxicity more than did optically light NOM in

rainbow trout. Studies generally agree that as pH increases the toxicity of Pb decreases (MacDonald et al., 2002; Horne and Dunson, 1995a,b,c). As pH decreases, lead becomes more soluble and more readily bioavailable to aquatic organisms (Weber, 1993). Acute and chronic toxicity of lead increases with decreasing water hardness, as lead becomes more soluble and bioavailable to aquatic organisms (Horne and Dunson, 1995c; Borgmann et al., 2005). There is some evidence that water hardness and pH work together to increase or decrease the toxicity of lead. High Ca^{2+} concentrations have been shown to protect against the toxic effects of lead (Sayer et al., 1989; Rogers and Wood, 2004; MacDonald et al., 2002; Hassler et al., 2004). Ca^{2+} affects the permeability and integrity of cell membranes and intracellular contents (Sayer et al., 1989). As Ca^{2+} concentrations decrease, the passive flux of ions (e.g., lead) and water increases. Finally, increasing salinity was found to decrease lead toxicity (Verslycke et al., 2003). The reduction in toxicity was attributed to increased complexation of Pb^{2+} with Cl^{-} ions.

Nutrients (e.g., nitrate, carbonate) have been shown to affect lead toxicity in some aquatic organisms. Jampani (1988) looked at the impact of various nutrients (i.e., sodium acetate, citric acid, sodium carbonate, nitrogen, and phosphates) on reducing growth inhibition in blue-green algae (*Synechococcus aeruginosus*) exposed to 200 mg/L of lead. Results indicated that additional nitrogen, phosphates, and some carbon sources, including sodium acetate, citric acid and sodium carbonate, all protected the algae from lead toxicity. One hypothesis was that nutrients were able to reverse toxic effects. The second hypothesis was that nutrients directly interacted with lead, in some way sequestering the metal so as to inhibit its metabolic interaction with the organism (Rao and Reddy, 1985; Jampani, 1988). Rai and Raizada (1989) investigated the effects of lead on nitrate and ammonium uptake and results indicated that lead exposure can affect the uptake of some nutrients in *N. muscorum*. Thus, nutrients seem to be capable of reducing toxicity, though the mechanisms have not been well established.

Interactions with Other Pollutants

Predicting the response of organisms to mixtures of chemicals is a daunting task (Norwood et al., 2003). Antagonism, synergism, and additivity are the primary responses that occur following exposure to multiple contaminants. When two or more metals compete for the same binding sites or interfere with transport through cell walls or membranes, the interaction is termed less than strictly additive or antagonistic. Antagonistic interactions can reduce metal

bioavailability when metals are present in combination, and may lead to reduced potential for toxicity (Hassler et al., 2004). There are a number of elements (Ca^{2+}, Cd^{2+}, Mg^{2+}, Na^{+} and Cl^{-}) that act in an antagonistic fashion with Pb (Niyogi and Wood, 2004; Rogers and Wood, 2003, 2004; Ahern and Morris, 1998; Li et al., 2004). For example, Pb is a well-known antagonist to Ca^{2+} (Hassler et al., 2004; Niyogi and Wood, 2004). Calcium is an essential element, required for a number of physiological processes in most organisms.

Synergism occurs when the interaction of two or more metals causes an effect that is greater than the effect observed from the individual metals themselves (Hagopian-Schlekat et al., 2001). Synergism is likely the result of increased bioavailability of one or more of the metal ions due to the presence of other metals (Hassler et al., 2004). Hassler et al. (2004) reported that in the presence of copper (Cu^{2+}) there was a significantly higher rate of internalization of Pb in the green algae *Chlorella kesserii*. It was suggested that Cu^{2+} may have affected organism physiology through the disruption of cell membrane integrity. This would allow increased cation (i.e., Pb^{2+}) permeability and therefore substantially increased internalization of Pb. Synergistic interactions have also been observed with lead and other metals (Cd, Cu, Ni, and Zn) (Hagopian-Schlekat et al., 2001).

The combined effects of two or more metals may result in additivity when the observed effects are greater than that observed with individual metals but equivalent to a summation of the effects from multiple metals. Norwood et al. (2003) reported that in a review and re-interpretation of published data on the interactions of metals in binary mixtures (n = 15 studies), antagonistic (6) and additive interactions (6) were the most common for lead. The two most commonly reported lead-element interactions are between lead and calcium and lead and zinc. Both calcium and zinc are essential elements in organisms and the interaction of Pb with these ions can lead to adverse effects both by increased Pb uptake and by a decrease in Ca and Zn required for normal metabolic functions.

8.2.1.4 Exposure/Response of Aquatic Species

Effects of Lead on Primary Producers

In the previous air quality criteria document (U.S. Environmental Protection Agency, 1986a), several authors reported that some algal species (e.g., *Scenedesmus sp.*) were found to exhibit physiological changes when exposed to high lead or organolead concentrations in situ.

The observed changes included increasing numbers of vacuoles, deformations in cell organelles, and increased autolytic activity. Increased vacuolization was assumed to be a tolerance mechanism by which lead was immobilized within cell vacuoles.

Several studies have been conducted since the 1986 air quality criteria document on the toxicity of lead to primary producers (Rai and Raizada, 1989; Jampani, 1988; Adam and Abdel-Basset, 1990; Gaur et al., 1994; Gupta and Chandra, 1994). Effects to algal growth (*Chlorella vulgaris, Closterium acerosum, Pediastrum simplex, Scenedesmus quadricauda*), ranging from minimal to complete inhibition, have been reported at lead concentrations between 100 and 200,000 µg/L (Bilgrami and Kumar, 1997; Jampani, 1988). The toxicity of lead to aquatic plant growth has been studied using *Spirodela polyrhiza*, *Azolla pinnata*, and *Lemna gibba* (Gaur et al., 1994; Gupta and Chandra, 1994; Miranda and Ilangovan, 1996). Test durations ranged from 4 to 25 days and test concentrations ranged between 49.7 and 500,000 µg/L (Gaur et al., 1994; Miranda and Ilangovan, 1996). Research on aquatic plants has been focussed on the effects of lead on aquatic plant growth, chlorophyll and protein content.

Algae and aquatic plants have a wide range in sensitivity to the effects of lead in water. Both groups of primary producers experience EC50 values for growth inhibition between approximately 1,000 and >100,000 µg/L (Bilgrami and Kumar, 1997; Jampani, 1988; Gaur et al., 1994). The most sensitive primary producers reported in the literature for effects to growth were *Closterium acersoum* and *Azolla pinnata* (Bilgrami and Kumar, 1997; Gaur et al., 1994). The least sensitive primary producers reported in the literature for effects to growth were *Synechococcus aeruginosus* and *Lemna gibba* (Jampani, 1988; Miranda and Ilangovan, 1996). Exposure to lead in combination with other metals is generally less toxic to growth than exposure to lead alone. Studies have shown that lead adversely affects the metabolic processes of nitrate uptake, nitrogen fixation, ammonium uptake, and carbon fixation (Rai and Raizada, 1989). Lead in combination with nickel or chromium produced synergistic effects for nitrate uptake, nitrogenase activities, ammonium uptake, and carbon fixation (Rai and Raizada, 1989).

Effects of Lead on Consumers

The 1986 AQCD (U.S. Environmental Protection Agency, 1986a) reported that hematological and neurological responses are the most commonly reported effects to aquatic vertebrates. These effects include red blood cell destruction and inhibition of the enzyme

ALAD, required for hemoglobin synthesis. The lowest reported exposure concentration causing either hematological or neurological effects was 8 µg/L (U.S. Environmental Protection Agency, 1986a).

Recent literature on the toxicity of lead to fish and aquatic invertebrates has been summarized by Eisler (2000). Exposure of invertebrates to Pb can lead to adverse effects on reproduction, growth, survival, and metabolism (Eisler, 2000). Water-borne lead is highly toxic to aquatic organisms, with toxicity varying depending on the species and life stage tested, duration of exposure, the form of lead tested, and water quality characteristics. Among the species tested, aquatic invertebrates, such as amphipods and water fleas, were the most sensitive to the effects of lead with adverse effects being reported at concentrations ranging from 0.45 to 8000 µg/L. Freshwater fish demonstrated adverse effects at concentrations ranging from 10 to >5400 µg/L, generally depending upon water quality parameters (e.g., pH, hardness, salinity). Amphibians tend to be relatively tolerant of lead, however, may exhibit decreased enzyme activity (e.g., ALAD reduction) and changes in behavior (e.g., hypoxia response behavior). Lead tends to be more toxic in longer-term exposures, with chronic toxicity thresholds for reproduction in water fleas ranging as low as 30 µg/L (e.g., Kraak et al., 1994).

8.2.1.5 Effects of Lead on Natural Aquatic Ecosystems

The effects of lead on natural aquatic ecosystems were examined following the conceptual framework developed by the EPA Science Advisory Board (Young and Sanzone, 2002). The essential attributes used to describe ecological condition include landscape condition, biotic condition, chemical and physical characteristics, ecological processes, hydrology and geomorphology and natural disturbance regimes. The majority of the published literature pertaining to lead and natural aquatic ecosystems focuses on the biotic condition and identifies effects on energy flow or nutrient cycling, community structure, and predator-prey interactions.

Recent studies have attributed the presence of lead to reduced primary productivity, respiration, and alterations of community structure. Specifically, lead (6-80 mg/L) was found to reduce primary productivity and increase respiration in an algal community (Jayaraj et al., 1992). Laboratory microcosm studies have indicated reduced species abundance and diversity in protozoan communities exposed to 0.02 – 1 mg/L (Fernandez-Leborans and Novillo, 1992, 1994; Fernandez-Leborans and Antonio-García, 1988). Numerous field studies have associated the

presence or bioaccumulation of lead with reductions of species abundance, richness, or diversity, particularly in benthic macroinvertebrate communities (Deacon et al., 2001; Mize and Deacon, 2002; Mucha et al., 2003; Poulton et al., 1995; Rhea et al., 2004; Maret et al., 2003). However, in natural aquatic ecosystems, lead is often found coexisting with other metals and other stressors. Thus, understanding the effects of lead in natural systems is challenging given that observed effects may be due to cumulative toxicity from multiple stressors.

Exposure to lead in laboratory studies and simulated ecosystems may alter species competitive behaviors, predator-prey interactions, and contaminant avoidance behaviors. Alteration of these interactions may have negative effects on species abundance and community structure. For example, reduced avoidance behaviors have been observed at lead concentrations ranging from 0.3 – 1.0 mg/L (Weber, 1996; Steele et al., 1991; Weis and Weis, 1998). The feeding behaviors of competitive species in some aquatic organisms are also influenced by the presence of lead (Lefcort et al., 2000).

The effects of lead have primarily been studied in instances of point source pollution rather than area-wide atmospheric deposition. Thus, the effects of atmospheric lead on aquatic ecological condition remains to be defined. There is a paucity of data in the general literature that explores the effects of lead in conjunction with all or several of the various components of ecological condition as defined by the EPA (Young and Sanzone, 2002). However, numerous studies are available associating the presence of lead with effects on biotic conditions.

8.2.2 Methodologies in Aquatic Ecosystem Research

8.2.2.1 Introduction

As discussed in previous sections, aerial deposition is one source of Pb deposition to aquatic systems. Consequently, to develop air quality criteria for Pb, consideration must be given to not only the environmental fate of Pb, but also to the environmental effects of Pb in the aquatic environment through consideration of laboratory toxicity studies and field evaluations. Perhaps the most straightforward approach for evaluating the effects of Pb is to consider extant criteria for Pb in aquatic ecosystems, i.e., water and sediment quality criteria. A key issue in developing Pb water and sediment criteria that are broadly applicable to a range of water bodies is properly accounting for Pb bioavailability and the range in species sensitivities. This section summarizes how these criteria are derived, the types of toxicity studies considered, and key

factors that influence the bioavailability of Pb in surface water and sediment to aquatic life. Because Pb in the aquatic environment is often associated with other metals (e.g., cadmium, copper, zinc), the importance of considering the toxicity of metal mixtures is also discussed. Finally, some issues related to background Pb concentrations are briefly addressed. It is beyond the scope of this section to review all methodologies in aquatic system research, but good reviews can be found in summary books, such as Rand et al. (1995).

8.2.2.2 Analytical Methods

Common analytical methods for measuring Pb in the aquatic environment are summarized in Table 8-2.2.1. For relevance to the ambient water quality criteria (AWQC) and sediment quality criteria for Pb discussed below, minimum detection limits should be in the low parts per billion (ppb) range for surface water and the low parts per million (ppm) range for sediment.

Table 8-2.2.1. Common Analytical Methods for Measuring Lead in Water, Sediment, and Tissue

Analysis Type	Analytical Method
Direct-Aspiration (Flame) Atomic Absorption Spectroscopy (AAS)	EPA SW-846 Method 7420[a], EPA Method 239.1[b], Standard Method 3111[c]
Graphite Furnace Atomic Absorption Spectroscopy (GFAAS)	EPA SW-846 Method 7421[a], EPA Method 239.2[b], Standard Method 3113[c]
Inductively Coupled Plasma (ICP)	EPA SW-846 Method 6010B[a], EPA Method 200.7[b], Standard Method 3120[c]
Inductively Coupled Plasma-Mass Spectrometry (ICP-MS)	EPA SW-846 Method 6020[a], EPA Method 200.8[b]

[a] U.S. Environmental Protection Agency (1986c) Test Methods for Evaluating Solid Waste, Physical/Chemical Methods (SW-846). Third Edition, September 1986; Final Updates I (7/92), IIA (8/93), II (9/94), IIB (1/95), III (12/96), IIIA (4/98), IIIB (11/04).

[b] U.S. Environmental Protection Agency (1991) Methods for the Determination of Metals in Environmental Samples. EPA/600/4-91-010. June 1991 (Supplement I, EPA/600/R-94-111, May 1994).

[c] American Public Health Association (1995) Standard Methods for the Examination of Water and Wastewater, 19th Edition. American Public Health Association, American Water Works Association, Water Pollution Control Federation.

8.2.2.3 Ambient Water Quality Criteria: Development

The EPA's procedures for deriving AWQC are described in Stephan et al. (1985) and are summarized here. With few exceptions, AWQC are derived based on data from aquatic toxicity studies conducted in the laboratory. In general, both acute (short term) and chronic (long term) AWQC are developed. Depending on the species, the toxicity studies considered for developing acute criteria range in length from 48 to 96 hours. Acceptable endpoints for acute AWQC development are mortality and/or immobilization, expressed as the median lethal concentration (LC_{50}) or median effect concentration (EC_{50}). For each species, the geometric mean of the acceptable LC_{50}/EC_{50} data is calculated to determine the species mean acute value (SMAV). For each genera, the geometric mean of the relevant SMAVs is then calculated to determine the genus mean acute value (GMAV). The GMAVs are then ranked from high to low, and the final acute value (FAV; the 5th percentile of the GMAVs, based on the four GMAVs surrounding the 5th percentile) is determined. Because the FAV is based on LC_{50}/EC_{50} values (which represent unacceptably high levels of effect), the FAV is divided by two to estimate a low-effect level. This value is then termed the acute criterion, or criterion maximum concentration (CMC). Based on the most recent AWQC document for Pb (U.S. Environmental Protection Agency, 1985), Table 8-2.2.2 shows the freshwater SMAVs and GMAVs for Pb, and the resulting freshwater CMC. Note that the freshwater AWQC are normalized for the hardness of the site water, as discussed further below in Section 8.2.2.4.

To develop chronic AWQC, acceptable chronic toxicity studies should encompass the full life cycle of the test organism, although for fish, early life stage or partial life cycle toxicity studies are considered acceptable. Acceptable endpoints include reproduction, growth and development, and survival, with the effect levels expressed as the chronic value, which is the geometric mean of the no-observed-effect concentration (NOEC)[2] and the lowest-observed-effect concentration (LOEC)[3]. Although a chronic criterion could be calculated as the 5th percentile of genus mean chronic values (GMCVs), sufficient chronic toxicity data are generally lacking, as is the case for Pb. Consequently, an acute-chronic ratio (ACR) is typically applied to

[2] The NOEC is the highest concentration tested that did not result in statistically significant effects relative to the control.

[3] The LOEC is the lowest concentration tested that resulted in statistically significant effects relative to the control.

Table 8-2.2.2. Development of Current Acute Freshwater Criteria for Lead (U.S. Environmental Protection Agency, 1985)[1]

Rank	Species	GMAV (µg/L)	SMAV (µg/L)
10	Midge (*Tanytarsus dissimilis*)	235,900	235,900
9	Goldfish (*Carassius auratus*)	101,100	101,100
8	Guppy (*Poecilia reticulata*)	66,140	66,140
7	Bbluegill (*Lepomis macrochirus*)	52,310	52,310
6	Fathead minnow (*Pimephales promelas*)	25,440	25,440
5	Brook trout (*Salvelinus fontinalis*)	4,820	4,820
4	Rainbow trout (*Oncorhynchus mykiss*)	2,448	2,448
3	Snail (*Aplexa hypnorum*)	1,040	1,040
2	Cladoceran (*Daphnia magna*)	447.8	447.8
1	Amphipod (*Gammarus pseudolimnaeus*)	142.6	142.6
		FAV = 67.54 µg/L	
		CMC = 33.77 µg/L	

[1] All values are normalized to a hardness of 50 mg/L (see Section 8.2.2.4).

the FAV to derive the chronic criterion. As the name applies, the ACR is the ratio of the acute LC_{50} to the chronic value, based on studies with the same species and in the same dilution water. For Pb, the final ACR is 51.29, which results in a final chronic value (FCV) of 1.317 µg/L (at a hardness of 50 mg/L).

Subsequent sections summarize some of the toxicity studies that meet the AWQC development guidelines, with an emphasis on key studies published since the last Pb AWQC were derived in 1984.

8.2.2.4 Ambient Water Quality Criteria: Bioavailability Issues

In surface waters, the environmental fate of metal contaminants is mitigated through adsorption, complexation, chelation, and other processes that affect bioavailability. The toxicity

of divalent cations tends to be highest in soft waters with low concentrations of dissolved organic matter and suspended particles. In an acidic environment (pH <4), the ionic form of most metals generally predominates and is considered to be the more toxic form. As the pH increases, carbonate, oxide, hydroxide, and sulfide complexes of the metals tend to predominate, and tend to be less toxic (Florence, 1977; Miller and Mackay, 1980). The portion of dissolved metal available for uptake or bioaccumulation is influenced by modifying factors that "sequester" the metal in an environmental matrix, thereby reducing the bioavailability of the metal at the sites of action. Metals can become complexed (bound) to a ligand that can make metals either more toxic (via transport mechanisms) or less toxic (by changing the metal's biological activity). Metals that complex tightly to ligands generally are not readily bioavailable and, thus, are less toxic to aquatic biota than their free-metal ion counterparts (Carlson et al., 1986; McCarthy, 1989). There are many kinds of ligands, organic and inorganic, as well as natural and man-made. Ligands found in natural surface waters and municipal and industrial effluent discharges include glycine, ammonia, oxalate, humic or fulvic acids, hydroxide, carbonate, bicarbonate, chloride, and hydrogen sulfide (Stumm and Morgan, 1970; Martin, 1986; Pagenkopf, 1986).

Recognizing the importance of calcium and magnesium ions (hardness) in modifying Pb toxicity, the current freshwater AWQC for Pb are normalized based on the hardness of the site water. The acute freshwater criteria, for example, are 34, 82, and 200 μg/L at hardness levels of 50, 100, and 200 mg/L (as $CaCO_3$). Although it has been known for some time that other water quality parameters such as pH, dissolved organic carbon (DOC), and alkalinity affect the bioavailability of metals to aquatic biota, it was the relatively recent development of the biotic ligand model (BLM) that allowed AWQC to consider all of these factors. Paquin et al. (2002) provided a thorough review of the factors influencing metal bioavailability and how research over the last few decades has culminated in the development of the BLM.

By understanding the binding affinities of various natural ligands in surface waters and how the freshwater fish gill interacts with free cations in the water, one can predict how metals exert their toxic effects (Schwartz et al., 2004). Early precursors to the BLM were the free-ion activity model (FIAM) and the gill surface interaction model (GSIM). The FIAM is a conceptual model that accounts for the binding of free metal ion and other metal complexes to the site of toxic action in an organism; it also considers competition between metal species and other cations (Paquin et al., 2002). The GSIM is fundamentally similar to the FIAM in that it accounts

for competition between metal ions and hardness cations at the physiological active gill sites, but whereas the FIAM is largely conceptual, the GSIM was used in interpreting toxicity test results for individual metals and metal mixtures (Pagenkopf, 1983). The BLM was adapted from the GSIM and uses the biotic ligand, rather than the fish gill as the site of toxic action (Di Toro et al., 2001; Paquin et al., 2002). This approach, therefore, considers that the external fish gill surface contains receptor sites for metal binding (Schwartz et al., 2004) and that acute toxicity is associated with the binding of metals to defined sites (biotic ligands) on or within the organism (Paquin et al., 2002). The model is predicated on the theory that mortality (or other toxic effects) occurs when the concentration of metal bound to biotic ligand exceeds a threshold concentration (Di Toro et al., 2001; Paquin et al., 2002). Direct uptake via the gills is thought to be the pathway for Pb uptake in freshwater fish (Merlini and Pozzi, 1977; Hodson et al., 1978). Free metal cations "out compete" other cations and bind to the limited number of active receptor sites on the gill surface, possibly suffocating and/or disrupting ionoregulatory mechanisms in the fish, leading to death (Di Toro et al., 2001; Paquin et al., 2002). Because the BLM uses the biotic ligand (not the fish gill) as the site of action, the model can be applied to other aquatic organisms, such as crustaceans, where the site of action is directly exposed to the aqueous environment (Di Toro et al., 2001). Dietary metals have also been shown to contribute to uptake by aquatic biota and, in some cases, increased toxicity. For example, Besser et al. (2005) observed that chronic (42-day) Pb toxicity to the amphipod *Hyalella azteca* was greater from a combined aqueous and dietary exposure than from a water-only exposure. The feasibility of incorporating dietary metals into BLMs is under investigation.

To date, the EPA has incorporated the BLM into draft freshwater criteria for copper, but the BLM is likely to be also included in the revised Pb criteria.

8.2.2.5 Sediment Quality Criteria: Development and Bioavailability Issues

As with metals in surface waters, the environmental fate of metal contaminants in sediments is moderated through various binding processes that reduce the concentration of free, bioavailable metal. Sediments function as a sink for Pb, as with most metals. Lead compounds such as Pb-carbonates, Pb-sulfates, and Pb-sulfides predominate in sediments (Prosi, 1989). Total Pb has a higher retention time and a higher percentage is retained in sediments compared to copper and zinc (Prosi, 1989). Lead is primarily accumulated in sediments as insoluble Pb

complexes adsorbed to suspended particulate matter. Naturally occurring Pb is bound in sediments and has a low geochemical mobility (Prosi, 1989). Organic-sulfide and moderately reducible fractions are less mobile, whereas cation-exchangeable fractions and easily-reducible fractions are more mobile and more readily bioavailable to biota (Prosi, 1989). Most Pb transported in surface waters is in a particulate form, originating from the erosion of sediments in rivers or produced in the water column (Prosi, 1989).

Sediment quality criteria have yet to be adopted by the EPA, but an equilibrium partitioning procedure has recently been published (U.S. Environmental Protection Agency, 2005c). The EPA has selected an equilibrium partitioning approach because it explicitly accounts for the bioavailability of metals. This approach is based on mixtures of cadmium, copper, Pb, nickel, silver, and zinc. Equilibrium partitioning (EqP) theory predicts that metals partition in sediment between acid-volatile sulfide, pore water, benthic organisms, and other sediment phases such as organic carbon. When the sum of the molar concentrations of simultaneously extracted metal (∑SEM) minus the molar concentration of AVS is less than zero, it can accurately be predicted that sediments are not toxic because of these metals. Note that this approach can be used to predict the lack of toxicity, but not the presence of toxicity. It is important to emphasize that metals must be evaluated as a mixture using this approach. If ndividual metals, or just two or three metals, are measured in sediment, ∑SEM would be misleadingly small and it may inaccurately appear that ∑SEM – AVS is less than 1.0.

If ∑SEM – AVS is normalized to the organic carbon fraction (i.e., (∑SEM – AVS)/f_{OC}), mortality can be more reliably predicted by accounting for both the site-specific organic carbon and AVS concentrations. When evaluating a metal mixture containing cadmium, copper, Pb, nickel, silver, and zinc, the following predictions can be made (U.S. Environmental Protection Agency, 2005c):

- A sediment with (SEM – AVS)/f_{OC} < 130 µmol/g_{OC} should pose low risk of adverse biological effects due to these metals.
- A sediment with 130 µmol/g_{OC} < (SEM – AVS)/f_{OC} < 3000 µmol/g_{OC} may have adverse biological effects due to these metals.
- In a sediment with (SEM – AVS)/f_{OC} > 3000 µmol/g_{OC}, adverse biological effects may be expected.

A third approach is to measure pore water concentrations of cadmium, copper, Pb, nickel, and zinc and then divide the concentrations by their respective FCVs. If the sum of these quotients is <1.0, these metals are not expected to be toxic to benthic organisms.

Many alternative approaches for developing sediment quality guidelines are based on empirical correlations between metal concentrations in sediment to associated biological effects, based on sediment toxicity tests (Long et al., 1995; Ingersoll et al., 1996; MacDonald et al., 2000). However, these guidelines are based on total metal concentrations in sediment and do not account for the bioavailability of metals between sediments. Sediment quality guidelines proposed for Pb from these other sources are shown in Table 8-2.2.3.

Table 8-2.2.3. Recommended Sediment Quality Guidelines for Lead

Source	Water Type	Guideline Type	Conc. (mg/kg dw)
MacDonald et al. (2000)	Freshwater	TEC	35.8
		PEC	128
Ingersoll et al. (1996)	Freshwater	ERL	55
		ERM	99
Long et al. (1995)	Saltwater	ERL	46.7
		ERM	218

TEC = Threshold effect concentration; PEC = Probable effect concentration; ERL = Effects range – low; ERM = Effects range – median

8.2.2.6 Metal Mixtures

As discussed above, the EPA's current approach for developing sediment criteria for Pb and other metals is to consider the molar sum of the metal concentrations (∑SEM). Although a similar approach has not been applied to AWQC, metal mixtures have been shown to be more toxic than individual metals (Spehar and Fiandt, 1986; Enserink et al., 1991). Spehar and Fiandt (1986) evaluated the acute and chronic toxicity of a metal mixture (arsenic, cadmium, chromium, copper, mercury, and Pb) to fathead minnows (*Pimephales promelas*) and a daphnid (*Ceriodaphnia dubia*). In acute tests, the joint toxicity of these metals was observed to be more than additive for fathead minnows and nearly strictly additive for daphnids. In chronic tests, the joint toxicity of the metals was less than additive for fathead minnows and nearly strictly

additive for daphnids. One approach for considering the additive toxicity of Pb with other metals is to use the concept of toxic units (TUs). Toxic units for each component of a metal mixture are derived by dividing metal concentrations by their respective acute or chronic criterion. The TUs for all the metals in the mixture are then summed. A $\sum TU > 1.0$ suggests the metal mixture is toxic (note that this is the same approach as discussed above for developing metal sediment criteria based on pore water concentrations). According to Norwood et al. (2003), the TU approach is presently the most appropriate model for predicting effects of metal mixtures based on the currently available toxicity data. However, it should also be emphasized that the TU approach is most appropriate at a screening level, because the true toxicity of the mixture is dependent on the relative amounts of each metal. The TU approach is also recommended with mixtures containing less than six metals.

For accessing Pb effects on aquatic ecosystems, it is not truly feasible to account for metal mixtures, because these will obviously vary highly from site to site. However, the toxicity of metal mixtures in surface water should be considered on a site-specific basis.

8.2.2.7 Background Lead

Because Pb is naturally occurring, it is found in all environmental compartments including surface water, sediment, and aquatic biota. Background Pb concentrations are spatially variable depending on geological features and local characteristics that influence Pb speciation and mobility. In the European Union risk assessments for metals, an "added risk" approach has been considered that assumes only the amount of metal added above background is relevant in a toxicological evaluation. However, this approach ignores the possible contribution of background metal levels to toxic effects, and background metal levels are regionally variable, precluding the approach from being easily transferable between sites. In terms of deriving environmental criteria for Pb, background levels should be considered on a site-specific basis if there is sufficient information that Pb concentrations are naturally elevated. As discussed previously, the use of radiogenic Pb isotopes is useful for source apportionment.

8.2.3 Distribution of Lead in Aquatic Ecosystems

8.2.3.1 Introduction

Atmospheric Pb is delivered to aquatic ecosystems primarily through deposition (wet or dry) or through erosional transport of soil particles (Baier and Healy, 1977; Dolske and Sievering, 1979). A number of physical and chemical factors govern the fate and behavior of Pb in aquatic systems. The EPA summarized some of these controlling factors in the 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a). For example, the predominant form of Pb in the environment is in the divalent (Pb^{2+}) form and complexation with inorganic and organic ligands is dependent on pH (Lovering, 1976; Rickard and Nriagu, 1978). A significant portion of Pb in the aquatic environment exists in the undissolved form (i.e., bound to suspended particulate matter). The ratio of Pb in suspended solids to Pb in filtrate varies from 4:1 in rural streams to 27:1 in urban streams (Getz et al., 1977). In still waters, Pb is removed through sedimentation at a rate determined by temperature, pH, oxidation-reduction (redox) potential, organic content, grain size, and chemical form of Pb in the water and biological activities (Jenne and Luoma, 1977). Since the publication of the 1986 Pb AQCD, knowledge of the properties of Pb in aquatic ecosystems has expanded. This section will provide further detail on the chemical species and the environmental factors affecting speciation of Pb in the aquatic environment. In addition, quantitative distributions of Pb in water, sediment, and biological tissues will be presented for aquatic ecosystems throughout the United States. Finally, recent studies discussing the tracing of Pb in aquatic systems will be summarized.

8.2.3.2 Speciation of Lead in Aquatic Ecosystems

The speciation of Pb in the aquatic environment is controlled by many factors. The primary form of Pb in aquatic environments is divalent (Pb^{2+}), while Pb^{4+} exists only under extreme oxidizing conditions (Rickard and Nriagu, 1978). Labile forms of Pb (e.g., Pb^{2+}, $PbOH^{+}$, $PbCO_3$) are a significant portion of the Pb inputs to aquatic systems from atmospheric washout. Lead is typically present in acidic aquatic environments as $PbSO_4$, $PbCl_4$, ionic Pb, cationic forms of Pb-hydroxide, and ordinary Pb-hydroxide ($Pb(OH)_2$). In alkaline waters, common species of Pb include anionic forms of Pb-carbonate ($Pb(CO_3)$) and $Pb(OH)_2$. Speciation models have been developed based on the chemical equilibrium model developed by Tipping (1994) to assist in examining metal speciation. The EPA MINTEQA2 computer model

(http://www.epa.gov/ceampubl/mmedia/minteq/) is one such equilibrium speciation model that can be used to calculate the equilibrium composition of dilute aqueous solutions in the laboratory or in natural aqueous systems. The model is useful for calculating the equilibrium mass distribution among dissolved species, adsorbed species, and multiple solid phases under a variety of conditions, including a gas phase with constant partial pressures. In addition to chemical equilibrium models, the speciation of metals is important from a toxicological perspective. The BLM was developed to study the toxicity of metal ions in aquatic biota and was previously described in Section 8.2.2.4. Further detail on speciation models is not provided herein, rather a general overview of major speciation principles are characterized in the following sections.

Acidity (pH)

Freshwater

Most of the Pb in aquatic environments is in the inorganic form (Sadiq, 1992). The speciation of inorganic Pb in freshwater aquatic ecosystems is dependent upon pH and the available complexing ligands. Solubility varies according to pH, temperature, and water hardness (Weber, 1993). Lead rapidly loses solubility above pH 6.5 (Rickard and Nriagu, 1978) and as water hardness increases. In freshwaters, Pb typically forms strong complexes with inorganic OH^- and CO_3^{2-} and weak complexes with Cl^- (Long and Angino, 1977; Bodek et al., 1988). The primary form of Pb at low pH ($\leq$6.5) is predominantly Pb^{2+} and less abundant inorganic forms include $Pb(HCO)_3$, $Pb(SO4)_2^{2-}$, PbCl, $PbCO_3$, and $Pb_2(OH)_2CO_3$ (Figure 8-2.3.1). At higher pH ($\geq$7.5), Pb forms hydroxide complexes ($PbOH^+$, $Pb(OH)_2$, $Pb(OH)_3^-$, $Pb(OH)_4^{2-}$).

Organic compounds in surface waters may originate from natural (e.g., humic or fulvic acids) or anthropogenic sources (e.g., nitrilotriacetonitrile and ethylenediaminetetraaceitc acid [EDTA]) (U.S. Environmental Protection Agency, 1986b). The presence of organic complexes has been shown to increase the rate of solution of Pb bound as Pb-sulfide (Lovering, 1976). Soluble organic Pb compounds are present at pH values near 7 and may remain bound at pH as low as 3 (Lovering, 1976; Guy and Chakrabarti, 1976). At higher pH (7.4 to 9), Pb-organic complexes are partially decomposed. Water hardness and pH were found to be important in Pb-humic acid interactions (O'Shea and Mancy, 1978). An increase in pH increased the concentration of exchangeable Pb complexes, while an increase in hardness tended to decrease

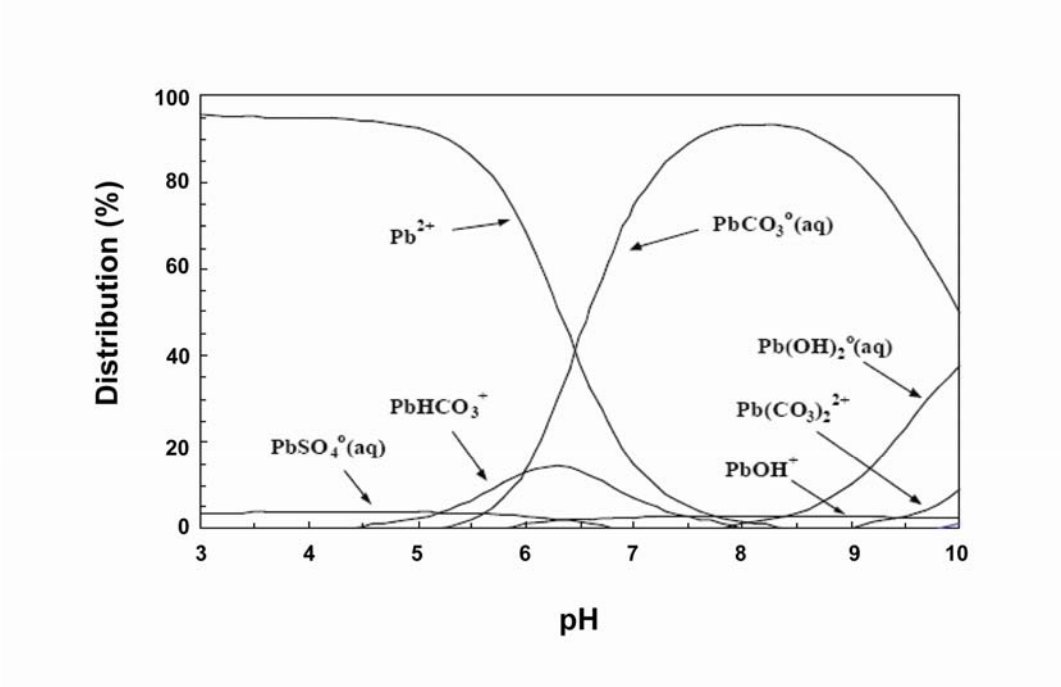

Figure 8-2.3.1. Distribution of lead aqueous species as a function of pH based on a concentration of 1 µg/L lead (U.S. Environmental Protection Agency, 1999).

the humic acid-Pb interactions. Thus, the metals involved in water hardness apparently inhibit the exchangeable interactions between metals and humic acids.

Marine Water

In marine systems, an increase in salinity increases complexing with chloride and carbonate ions and reduces the amount of free Pb^{2+}. In seawaters and estuaries at low pH, Pb is primarily bound to chlorides (PbCl, $PbCl_2$, $PbCl_3^-$, $PbCl_4^{2-}$) and may also form inorganic $Pb(HCO)_3$, $Pb(SO4)_2^{2-}$, or $PbCO_3$. Elevated pH in saltwater environments results in the formation of Pb hydroxides ($PbOH^+$, $Pb(OH)_2$, $Pb(OH)_3^-$, $Pb(OH)_4^{2-}$) (Figure 8-2.3.2). A recent examination of Pb species in seawater as a function of chloride concentration suggested that the primary species were $PbCl_3^- > PbCO_3 > PbCl_2 > PbCl^+$ > and $Pb(OH)^+$ (Fernando, 1995). Lead in freshwater and seawater systems is highly complexed with carbonate ligands suggesting that Pb is likely to be highly available for sorption to suspended materials (Long and Angino, 1977). Current information suggests that inorganic Pb is the dominant form in seawater; however, it has

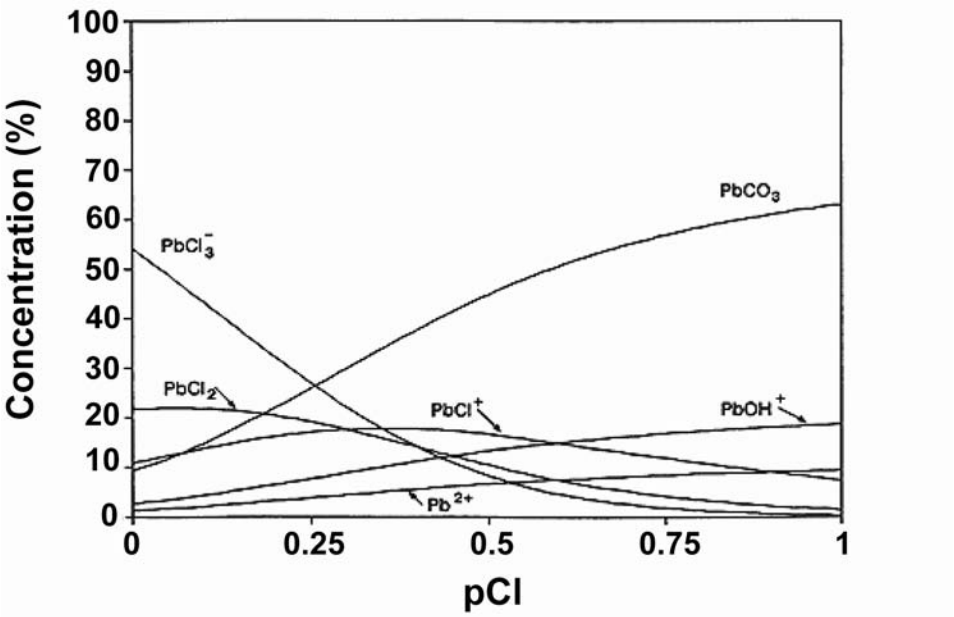

Figure 8-2.3.2. Lead speciation versus chloride content (Fernando, 1995).

been shown that organically bound Pb complexes make up a large portion of the total Pb (Capodaglio et al., 1990).

Sorption

Sorption processes (i.e., partitioning of dissolved Pb to suspended particulate matter or sediments) appear to exert a dominant effect on the distribution of Pb in the environment (U.S. Environmental Protection Agency, 1979). Sorption of Pb results in the enrichment of bed sediments, particularly in environments with elevated organic matter content from anthropogenicsources. Lead adsorption to aquatic sediments is correlated with pollution in sites containing high levels of anthropogenic organic content, even under acidic conditions (Tada and Suzuki, 1982; Brook and Moore, 1988; Davis and Galloway, 1993; Botelho et al., 1994; Davis et al., 1996). Particulate-bound forms are more often linked to urban runoff and mining effluents (Eisler, 2000).

Solid Pb complexes form when Pb precipitates or adsorbs to suspended particulate matter and sediments. Inorganic Pb adsorption to suspended organic matter or sediments is dependent on parameters such as, pH, salinity, water hardness, and the composition of the organic matter (U.S. Environmental Protection Agency, 1979). In addition to suspended organic matter, Pb can

adsorb to biofilms (i.e., bacteria) (Nelson et al., 1995; Wilson et al., 2001). Adsorption typically increases with increasing pH, increasing amounts of iron or manganese; and with a higher degree of polarity of the particulate matter (e.g., clays). Adsorption decreases with water hardness (Syracuse Research Corporation., 1999). At higher pH, Pb precipitates as $Pb(OH)^+$ and $PbHCO_3^+$ into bed sediments (Weber, 1993). Conversely, at low pH, Pb is negatively sorbed (repelled from the adsorbent surface) (U.S. Environmental Protection Agency, 1979; Gao et al., 2003). In addition, Pb may be remobilized from sediment with a decrease in metal concentration in the solution phase, complexation with chelating agents (e.g., EDTA), and changing redox conditions (Gao et al., 2003). Changes in water chemistry (e.g., reduced pH or ionic composition) can cause sediment Pb to become remobilized and potentially bioavailable to aquatic organisms (Weber, 1993).

Biotransformation

Methylation may result in Pb remobilization and reintroduction into the aqueous environment compartment and its subsequent release into the atmosphere (Syracuse Research Corporation., 1999). However, methylation is not a significant environmental pathway controlling the fate of Pb in the aquatic environment. The microbial methylation of Pb in aquatic systems has been demonstrated experimentally, but evidence for natural occurrence is limited (Beijer and Jernelov, 1984; DeJonghe and Adams, 1986). Reisinger et al. (1981) examined the methylation of Pb in the presence of numerous bacteria known to alkylate metals and did not find evidence of Pb methylation under any test condition. Tetramethyl-Pb may be formed by the methylation of Pb-nitrate or Pb-chloride in sediments (Bodek et al., 1988). However, tetramethyl-Pb is unstable and may degrade in aerobic environments after being released from sediments (U.S. Environmental Protection Agency, 1986b). Methylated species of Pb may also be formed by the decomposition of tetralkyl-Pb compounds (Radojevic and Harrison, 1987; Rhue et al., 1992). Sadiq (1992) reviewed the methylation of Pb compounds and suggested that chemical methylation of Pb is the dominant process and that biomethylation is of secondary importance.

8.2.3.3 Spatial Distribution of Lead in Aquatic Ecosystems

The 1986 Pb AQCD did not describe the distribution and concentration of Pb throughout aquatic ecosystems of the United States. Consequently, an analysis of readily available data on Pb concentrations was conducted to determine the distribution of Pb in the aquatic environment. Data from the United States Geological Survey (USGS) National Water-Quality Assessment (NAWQA) program were queried and retrieved. NAWQA contains data on Pb concentrations in surface water, bed sediment, and animal tissue for more than 50 river basins and aquifers throughout the country, and it has been used by the EPA for describing national environmental concentrations for use in developing AWQC.

NAWQA data are collected during long-term, cyclical investigations wherein study units undergo intensive sampling for 3 to 4 years, followed by low-intensity monitoring and assessment of trends every 10 years. The NAWQA program's first cycle was initiated in 1991; therefore, all available data are less than 15 years old. The second cycle began in 2001 and is ongoing; data are currently available through 30 September 2003. The NAWQA program study units were selected to represent a wide variety of environmental conditions and contaminant sources; therefore, agricultural, urban, and natural areas were all included. Attention was also given to selecting sites covering a wide variety of hydrologic and ecological resources.

NAWQA sampling protocols are designed to promote data consistency within and among study units while minimizing local-scale spatial variability. Water-column sampling is conducted via continuous monitoring, fixed-interval sampling, extreme-flow sampling, as well as seasonal, high-frequency sampling in order to characterize spatial, temporal, and seasonal variability as a function of hydrologic conditions and contaminant sources. Sediment and tissue samples are collected during low-flow periods during the summer or fall to reduce seasonal variability. Where possible, sediment grab samples are collected along a 100-m stream reach, upstream of the location of the water-column sampling. Five to ten depositional zones at various depths, covering left bank, right bank, and center channel, are sampled to ensure a robust representation of each site. Fine-grained samples from the surficial 2 to 3 cm of bed sediment at each depositional zone are sampled and composited. Tissue samples are collected following a National Target Taxa list and decision trees that help guide selection from that list to accommodate local variability.

The NAWQA dataset was chosen over other readily available national databases (i.e. the USEPA-maintained database for the STOrage and RETrieval [STORET] of chemical, physical, and biological data), because the study design and methods used to assess the water quality of each study unit are rigorous and consistent, and, as such, these data may be presented with a high level of confidence. This is in stark contrast to the STORET database, which essentially serves as a depot for any organization wishing to share data they have generated. This lack of a consistent methodology or QA/QC protocol has lead to the STORET data being highly qualified and offered with only a mild level of confidence. Furthermore, because there is no standard for site selection within STORET, the database may be biased toward contaminated sites. Finally, and, perhaps most importantly, the majority of the available Pb data in STORET predate the use of clean techniques for Pb quantification.[4]

Data Acquisition and Analysis

The following data were downloaded for the entire United States (all states) from the NAWQA website (http://water.usgs.gov/nawqa/index.html): site information, dissolved Pb concentration in surface water (µg/L), total Pb concentration (µg/g) in bed sediment (<63µm)[5], and Pb concentration in animal tissue (µg/g dw). Using the land use classification given for each site, the data were divided into two groups: "natural" and "ambient" (Table 8-2.3.1). All samples were considered to fall within the ambient group (the combined contribution of natural and anthropogenic sources), whereas the natural group comprised "forest," "rangeland," or "reference" samples only[6]. These groups follow those defined and recommended for use by the EPA's Framework for Inorganic Metals Risk Assessment (U.S. Environmental Protection Agency, 2004c). Finally, in addition to the natural/ambient classification, tissue samples were further divided into "whole organism" and "liver" groups.

[4] The authors recognize the existence of several local and regional datasets that may be of quality equal to NAWQA; however, due to the national scope of this assessment, these datasets were not included in the following analyses. We were unable to identify any monitoring data of similar quality for the marine/estuarine environment.

[5] NAWQA sediment samples are sieved to <63µm to promote the collection of fine-grained surficial sediments, which are natural accumulators of trace elements.

[6] The authors acknowledge that while Pb samples collected from sites classified under these three land use categories will most closely reflect natural background concentrations, atmospheric input of lead may be present.

Table 8-2.3.1. NAWQA Land Use Categories and Natural/Ambient Classification

NAWQA Land Use Categories	Classification
Agricultural	Ambient
Commercial/Industrial	Ambient
Cropland	Ambient
Forest	Ambient/Natural
Mining	Ambient
Mixed	Ambient
NA	Ambient
Orchard/Vineyard	Ambient
Other/Mixed	Ambient
Pasture	Ambient
Rangeland	Ambient/Natural
Reference	Ambient/Natural
Residential	Ambient
Urban	Ambient

All data were compiled in spreadsheets wherein non-detect values were converted to one-half of the detection limit and the total number of samples, percentage of non-detect values(percent censorship), minimum, maximum, median, standard deviation, and cumulative density functions were calculated for each endpoint for both the natural and ambient groups. Since all data were geo-referenced, a geographic information system (GIS; ArcGIS) was used to generate maps, conduct spatial queries and analyses, and calculate statistics.

Lead Distributions Generated from the NAWQA Database

Natural versus Ambient Groups

There were four to eight times more ambient surface water (Table 8-2.3.2) and bulk sediment (Table 8-2.3.3) samples in the compiled dataset than natural samples. This is most likely a function of both the NAWQA program site selection process and the fact that sites unaffected by human activities are extremely limited. The spatial distributions of natural and

Table 8-2.3.2. Summary Statistics of Ambient and Natural Levels of Dissolved Lead in Surface Water

	Surface Water Dissolved Pb (µg/L)	
Statistic	**Natural**	**Ambient**
% Censorship	87.91	85.66
N	430	3445
Minimum	0.04	0.04
Maximum	8.40	29.78
Mean	0.52	0.66
Standard Deviation	0.59	1.20
95th Percentile	0.50	1.10
96th Percentile	0.67	2.00
97th Percentile	1.00	2.34
98th Percentile	1.79	3.58
99th Percentile	2.48	5.44

Table 8-2.3.3. Summary Statistics of Ambient and Natural Levels of Total Lead in <63 µm Bulk Sediment

	Bulk Sediment <63 µm Total Lead (µg/g)	
Statistic	**Natural**	**Ambient**
% Censorship	1.16	0.48
N	258	1466
Minimum	0.50	0.50
Maximum	12000	12000
Mean	109.07	120.11
Standard Deviation	786.74	672.41
Median	22.00	28.00
95th percentile	161.50	200.00

ambient surface water/sediment sites were fairly comparable, with natural samples located in almost all of the same areas as ambient samples except in the Midwest (Ohio, Illinois, Iowa, and Michigan), where natural sites were not present (Figure 8-2.3.3). This exception may be because these areas are dominated by agricultural and urban areas. The same spatial distributions were observed for the natural and ambient liver and whole organism tissue samples (Figure 8-2.3.4 and Figure 8-2.3.5).

Surface Water

The total number of surface water Pb samples was 3,445; however these data were highly censored with 85.66% of the ambient samples (2951/3445) and 87.91% of the natural samples (378/430) below the detection limit[7] (Table 8-2.3.2). Consequently, the majority of the variability between these two datasets fell between the 95th and 100th (maximum) percentiles, as was shown by the frequency distributions of the two groups deviating only at the upper and lower tails with most of the overlapping data falling at 0.50 µg/L (one-half of the most common detection limit, 1.0 µg/L; Figure 8-2.3.6). As expected, due to the definitions of the natural and ambient groups, the 95th and 100th percentiles were consistently higher for the ambient samples than the natural samples. Similarly, the mean ambient Pb concentration (0.66 µg/L) was higher than the mean natural Pb concentration (0.52 µg/L).[8]

Due to the preponderance of non-detectable (ND) measurements, assessing national trends in surface water-dissolved Pb concentrations was not possible. However, areas with elevated Pb concentrations were identified by classifying the data with detectable Pb concentrations above and below the 99th percentile[9]. Areas with high surface water Pb concentrations were observed in Washington, Idaho, Utah, Colorado, Arkansas, and Missouri (Figure 8-2.3.7). The maximum

[7] The NAWQA dataset contains multiple detection limits for Pb in surface water. While the majority of data were analyzed with a detection limit of 1.0 µg/L (before 2000/2001), the most recent samples were analyzed with either a 0.5, 0.2, 0.16, or 0.08 µg/L detection limit (after 2000/2001), and some older samples (N = 20) were analyzed with a detection limit of 2.0 µg/L.

[8] The same pattern was observed upon calculating the mean Pb concentrations based on detect data only (ambient mean = 1.66 µg/L, natural mean = 0.87 µg/L); however, deletion of non-detect data has been shown to increase the relative error in the mean to a greater extent than inclusion of non-detects as ½ of the detection limit (Newman et al., 1989); therefore means were calculated using the latter method for this analysis.

[9] The 99th percentile (versus the 95th percentile) was chosen in this instance to represent extreme conditions given the small window of variability in the dataset. The 95th percentile will be used in subsequent analyses of this type.

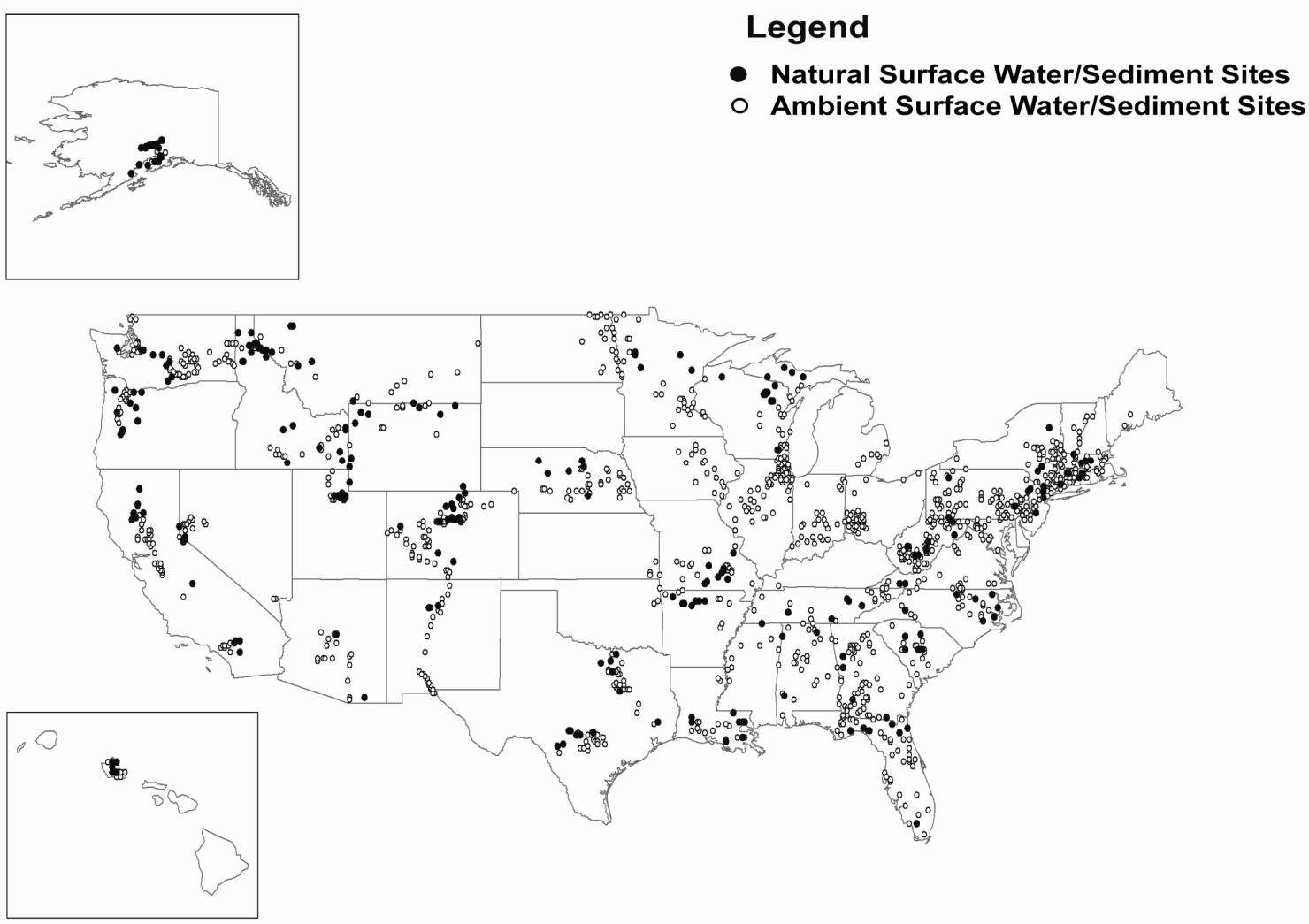

Figure 8-2.3.3. Spatial distribution of natural and ambient surface water/sediment sites (Surface water: natural N = 430, ambient N = 3445; Sediment: natural N = 258, ambient N = 1466).

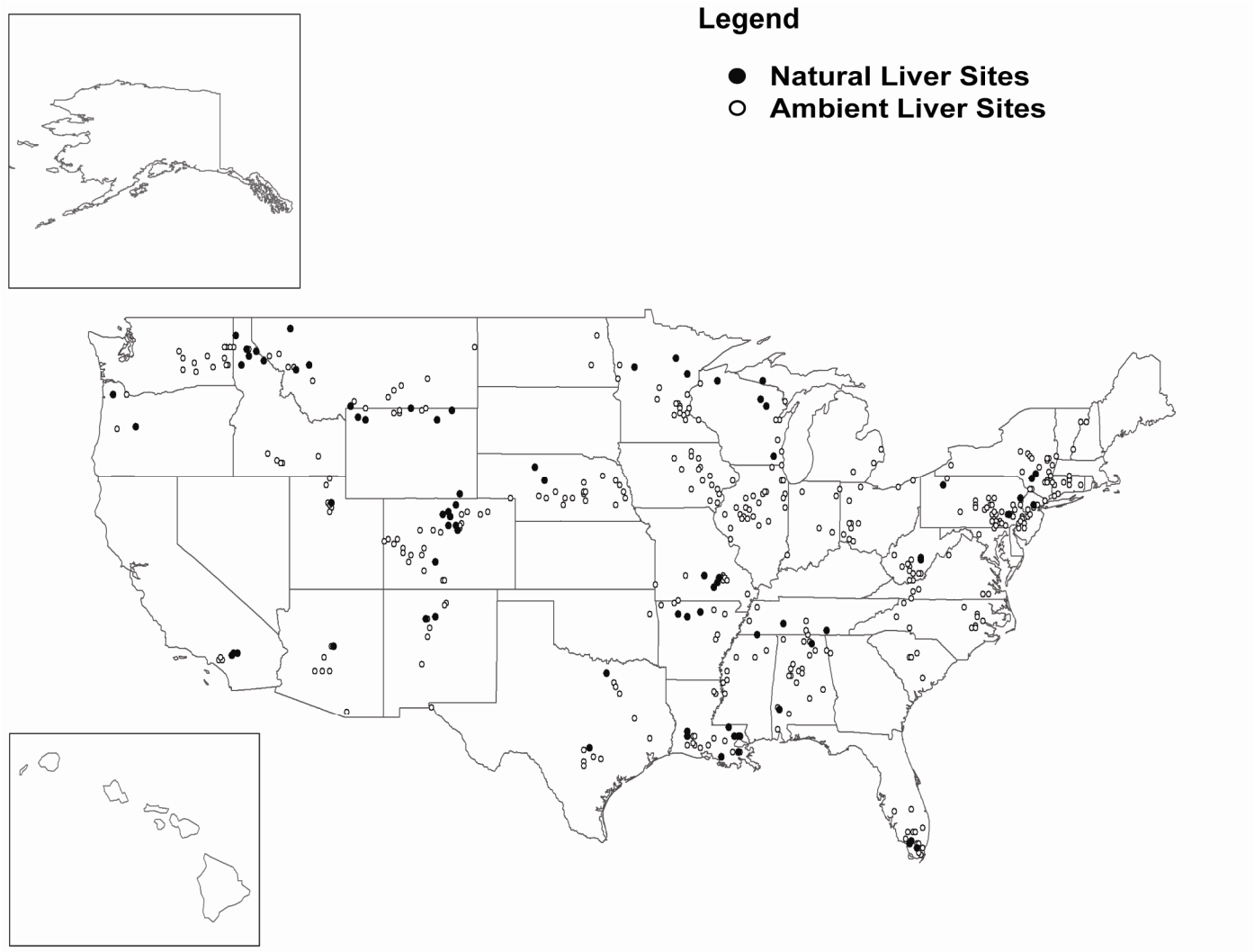

Figure 8-2.3.4. Spatial distribution of natural and ambient liver tissue sample sites (Natural N = 83, Ambient N = 559).

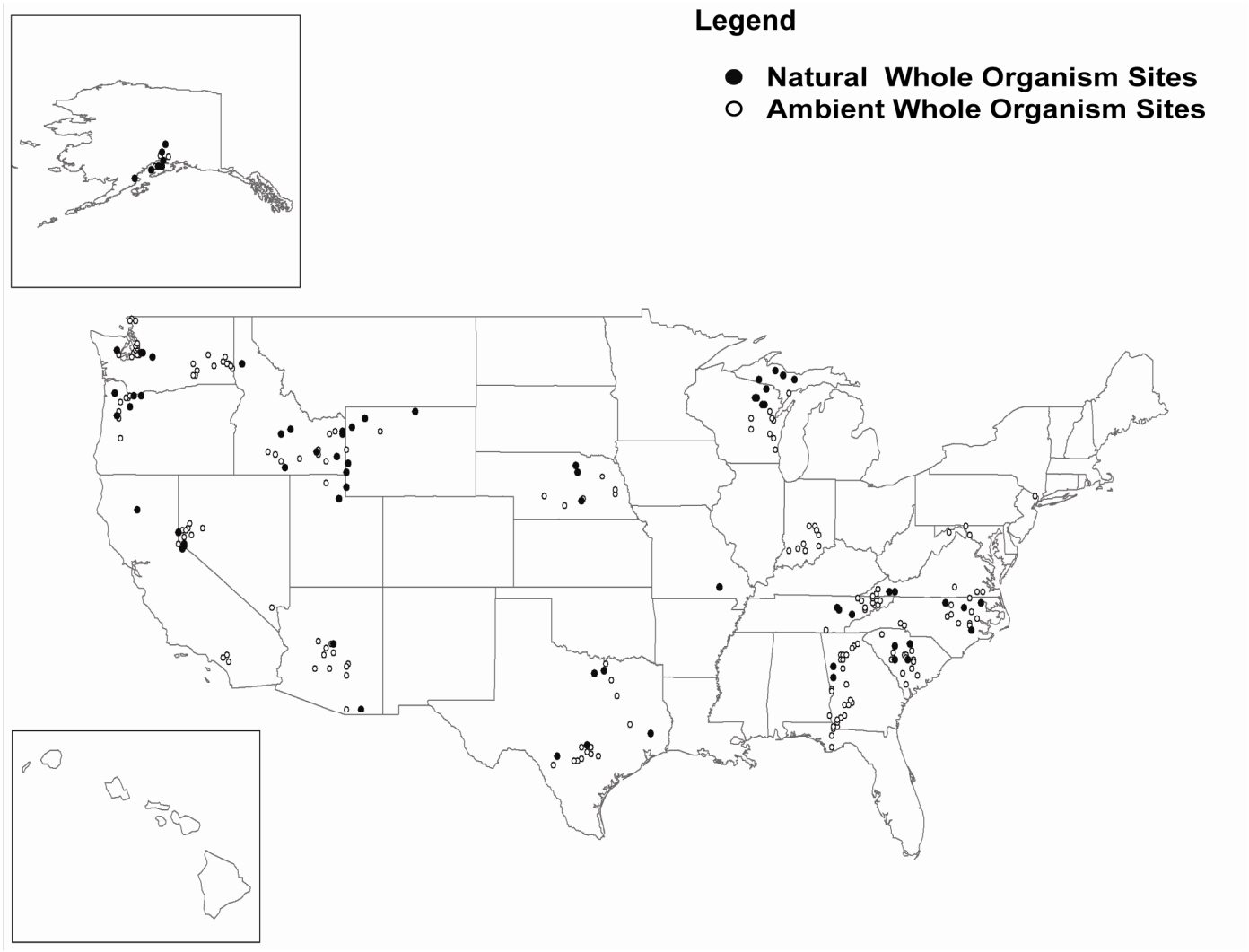

Figure 8-2.3.5. Spatial distribution of natural and ambient whole organism tissue sample sites (Natural N = 93, Ambient N = 332).

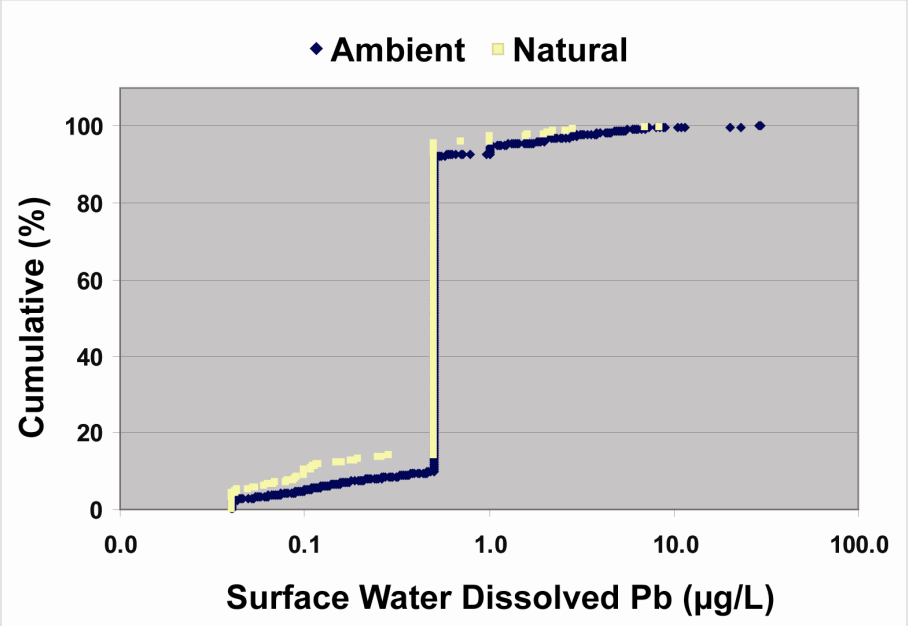

Figure 8-2.3.6. Frequency distribution of ambient and natural levels of surface water dissolved lead (µg/L).

measured Pb concentration was located in Canyon Creek at Woodland Park, ID, a site classified as mining land use.

Sediment

There were approximately one-half of the number of surface water data available for sediments (N = 1466). In contrast to the surface water data, however, very few sediment data were below the detection limit (7/1466 ambient ND, 3/258 natural ND; Table 8-2.3.3). As expected, the mean ambient Pb concentration was higher than the mean natural Pb concentration (120.11 and 109.07 µg/g, respectively). Similarly, the median ambient Pb concentration was higher than the median natural Pb concentration (28.00 and 22.00 µg/g, respectively) and the ambient 95th percentile was higher than the natural 95th percentile (200.00 and 161.50 µg/g, respectively). While the natural and ambient surface water Pb distributions differed only at the extremes, the natural sediment Pb percentiles were consistently lower than the ambient percentiles throughout the distributions (Figure 8-2.3.8). Unlike the

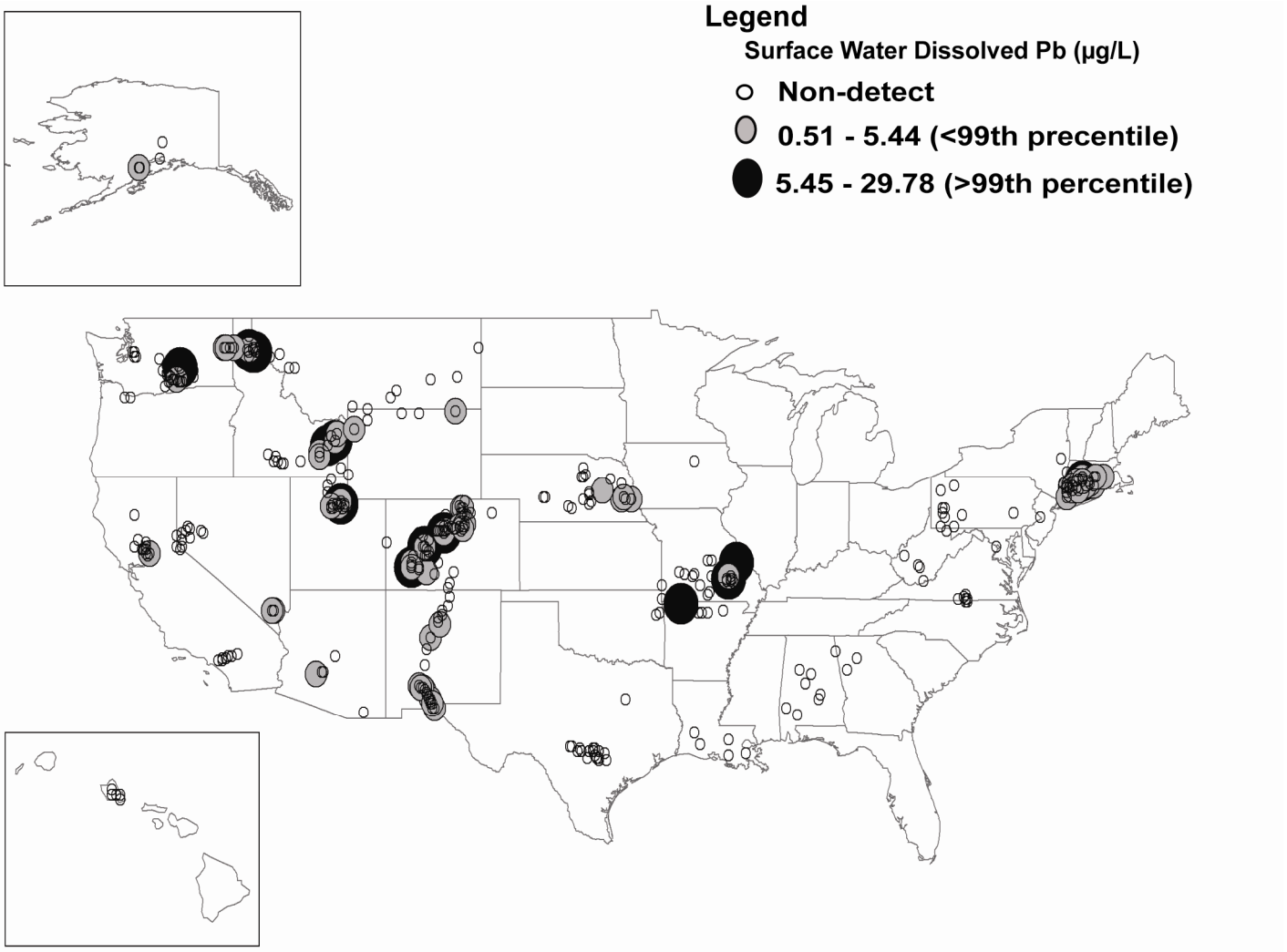

Figure 8-2.3.7. Spatial distribution of dissolved lead in surface water (N = 3445).

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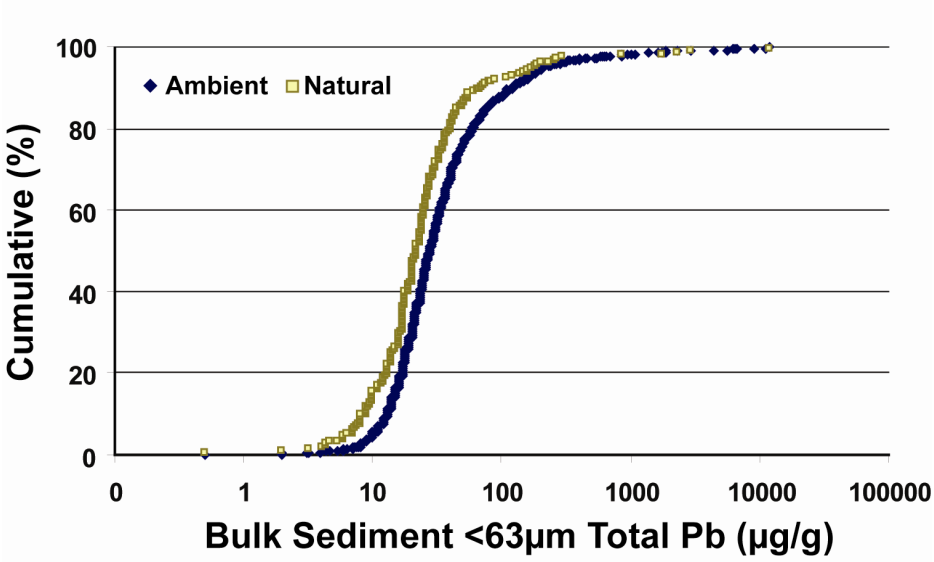


Figure 8-2.3.8. Frequency distribution of ambient and natural levels of bulk sediment <63 µm total Pb (µg/g).

surface water dataset, because the sediment dataset was not heavily censored, assessing national trends in sediment Pb concentrations was possible. The data were mapped and categorized into the four quartiles of the frequency distribution (Figure 8-2.3.9). The following observations were made:

- Sediment Pb concentrations generally increased from west to east (the majority of sites along East Coast had Pb concentrations in the fourth quartile of the sediment Pb concentration frequency distribution).

- Several "hot spots" of concentrated sites with elevated sediment Pb concentrations were apparent in various western states.

- Sediment Pb concentrations were generally lowest in the midwestern states (the majority of sites in North Dakota, Nebraska, Minnesota, and Iowa had Pb concentrations in the first or second quartile of sediment Pb concentration frequency distribution).

As was seen with surface water Pb concentrations, the highest measured sediment Pb concentrations were found in Idaho, Utah, and Colorado. Not surprisingly, of the top 10 sediment Pb concentrations recorded, 7 were measured at sites classified as mining land use.

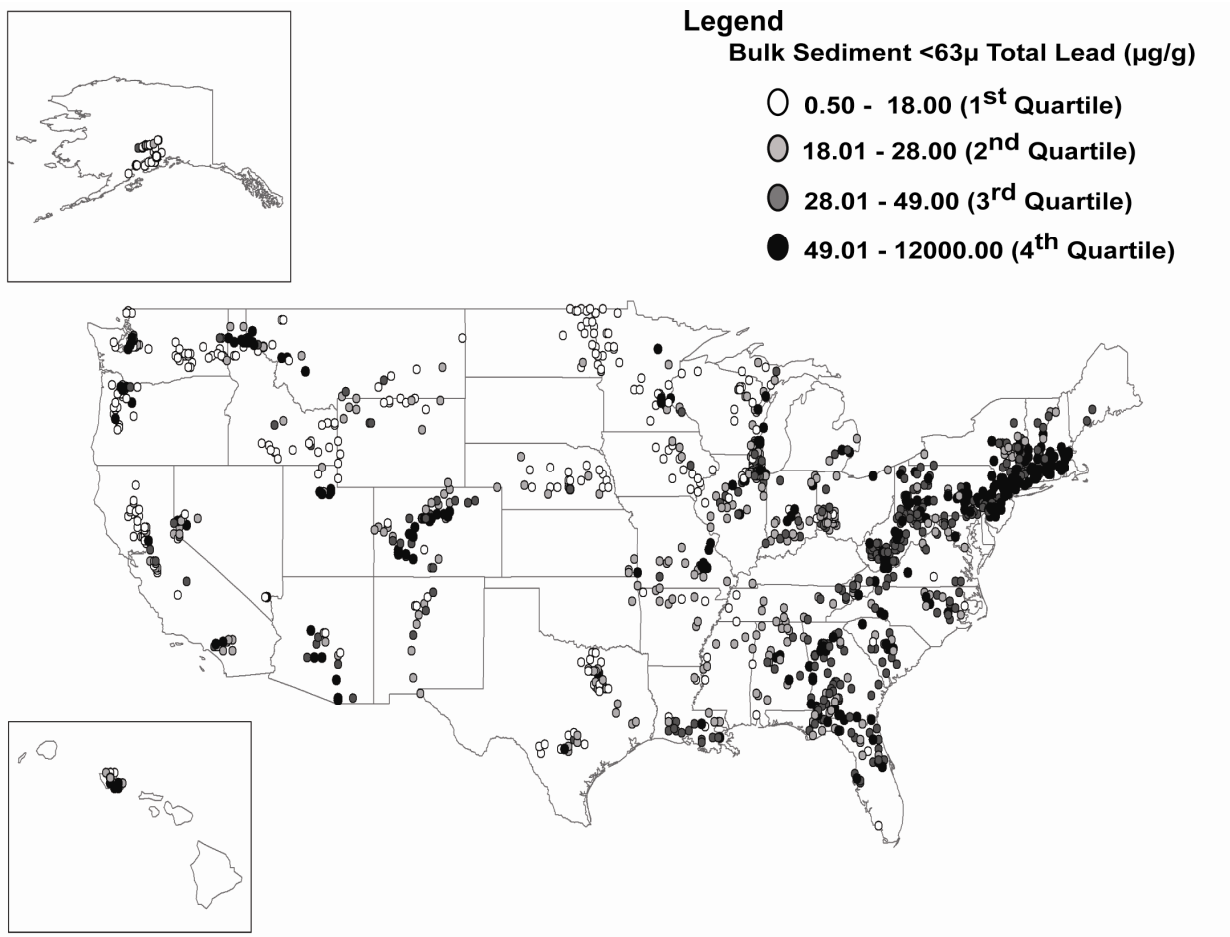


Figure 8-2.3.9. Spatial distribution of total lead in bulk sediment <63 µm (N = 1466).

Tissue

As was true for the surface water data, there were a high number of tissue samples below the detection limit (47/93 natural whole organism ND, 130/332 ambient whole organism ND, 74/83 natural liver ND, 398/559 ambient liver ND; Table 8-2.3.4). In general, more non-censored data were available for whole organism samples than liver samples, and for ambient sites than natural sites. As expected, for whole organism samples, the 95th percentile Pb concentration measured at ambient sites was higher than that measured at natural sites (3.24 and 2.50 µg/g, respectively); however, Pb liver concentration 95th percentiles for ambient and natural samples were very similar, with the natural 95th percentile actually higher than the ambient 95th percentile (1.26 and 1.06 µg/g, respectively). In addition, as expected, the median and mean Pb liver concentrations of ambient samples (0.15 and 0.36 µg/g, respectively) were higher than the median and mean Pb liver concentrations of natural samples (0.11 and 0.28 µg/g, respectively). The same pattern was observed in the whole organism median and mean Pb concentrations (ambient: median = 0.59, mean = 1.03; natural: median = 0.35, mean = 0.95 µg/g). In addition, the frequency distributions of the liver and whole organism Pb concentrations followed the same trends, with the natural percentiles consistently lower than the ambient percentiles throughout the distributions (Figure 8-2.3.10 and Figure 8-2.3.11).

These whole organism results were compared with findings from the 1984 U.S. Fish and Wildlife Service (USFWS) National Contaminant Biomonitoring Program (NCBP) (Schmitt and Brumbaugh, 1990). As part of this program, 321 composite samples of 3 to 5 whole, adult fish of a single species were collected from 109 river and Great Lake stations throughout the country. Samples were analyzed for Pb concentrations (µg/g ww) and the geometric mean, maximum, and 85th percentile were calculated. Upon comparing these summary statistics with the equivalent NAWQA ambient group value (NCBP stations were representative of both natural and anthropogenically influenced conditions), a very strong agreement between the two analyses was observed for each endpoint (Table 8-2.3.5). For example, NCBP and NAWQA geometric mean Pb concentrations were nearly identical[10] (0.55 and 0.54 µg/g dw, respectively) and the 85th percentiles only differed by 0.5 µg Pb/g dw (NCBP, 1.10 and NAWQA, 1.60).

[10] The authors acknowledge that a high degree of censorship is present in both datasets and no firm conclusions can be drawn by comparing these means. The objective of this exercise was limited to showing how the NAWQA data compare to other national datasets.

Table 8-2.3.4. Summary Statistics of Ambient and Natural Levels of Lead in Whole Organism and Liver Tissues

	Tissue Pb (µg/g dry weight)			
	Whole Organism		Liver	
	Natural	Ambient	Natural	Ambient
% Censorship	50.54	39.16	89.16	71.20
N	93	332	83	559
Minimum	0.08	0.08	0.01	0.01
Maximum	22.60	22.60	3.37	12.69
Mean	0.95	1.03	0.28	0.36
Standard Deviation	2.53	1.74	0.54	0.96
Median	0.11	0.15	0.35	0.59
95th percentile	1.26	1.06	2.50	3.24

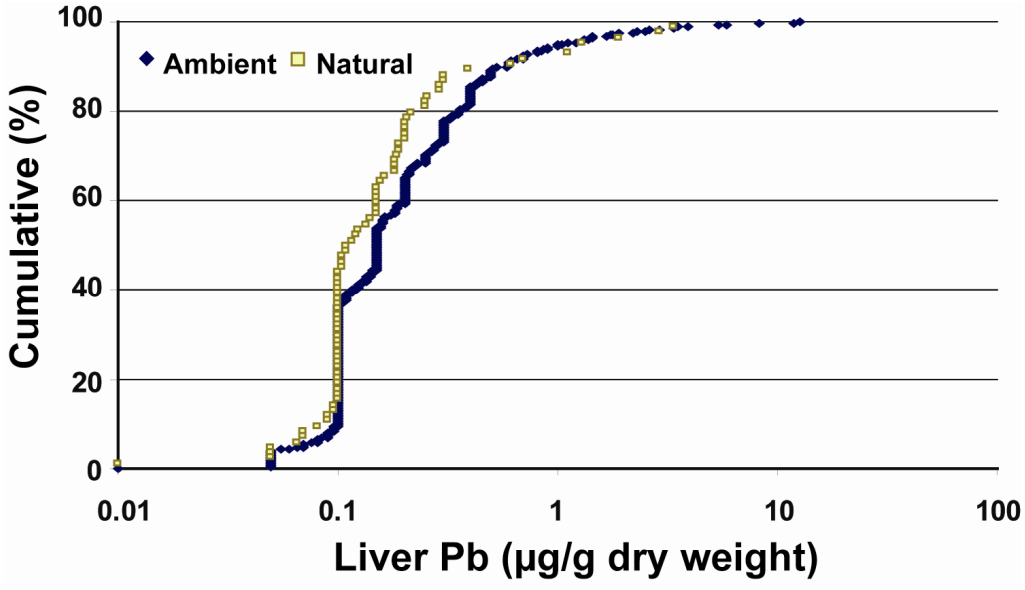


Figure 8-2.3.10. Frequency distribution of ambient and natural levels of lead in liver tissue (µg/g dry weight).

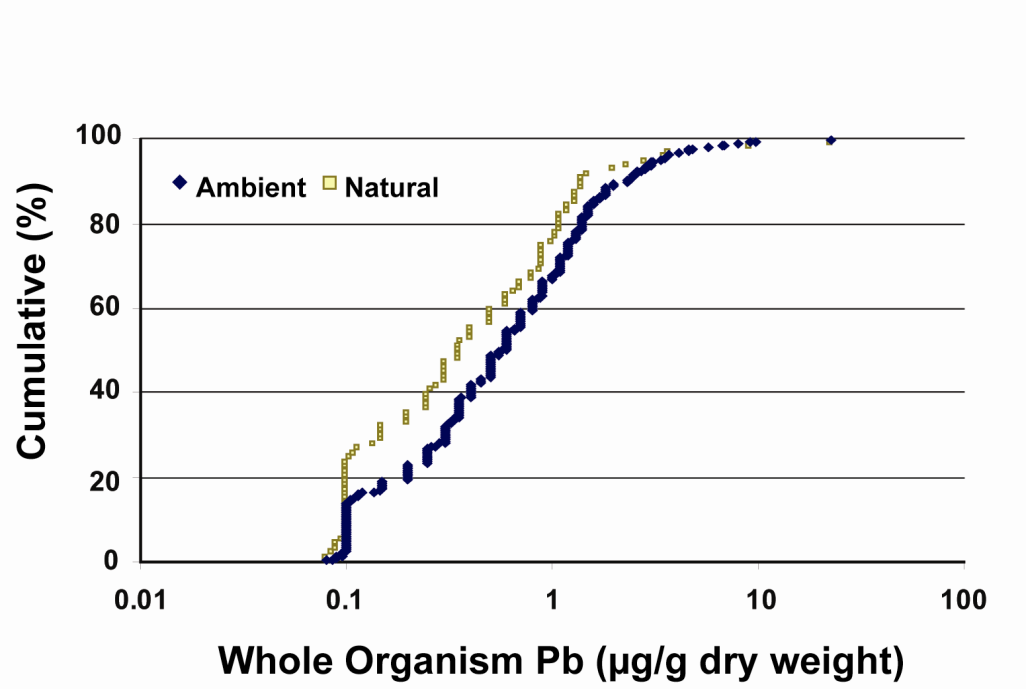


Figure 8-2.3.11. Frequency distribution of ambient and natural levels of lead in whole organism tissue (µg/g dry weight).

Table 8-2.3.5. Comparison of NCBP and NAWQA Ambient Lead Levels in Whole Organism Tissues

	Whole Organism Lead Concentration (µg/gd ry weight)	
Statistic	NCBP[1]	NAWQA
Geometric Mean	0.55	0.54
Maximum	24.40	22.60
85th Percentile	1.10	1.60

[1] To convert between wet and dry weight, wet weight values were multiplied by a factor of five.

As was the case with surface water data, the high amount of non-detectable measurements did not allow for a national assessment of spatial trends in Pb tissue concentrations. Instead, areas with high Pb tissue concentrations were identified by classifying the data above and below

the 95th percentile. Similar to surface water and sediments, tissue concentrations were found to be elevated in Washington, Idaho, Utah, Colorado, Arkansas, and Missouri; however, several of the highest measured Pb concentrations were also found in study units in the southwestern and southeastern states (Figure 8-2.3.12 and Figure 8-2.3.13). As expected, the majority of the samples with elevated Pb concentrations were taken from sites classified as urban, commercial/industrial, or mining.

8.2.3.4 Tracing the Fate and Transport of Lead in Aquatic Ecosystems

The following section presents a generalized framework for the fate and transport of Pb in aquatic systems (Figure 8-2.3.14). The primary source of Pb in natural systems is atmospheric deposition (Rickard and Nriagu, 1978; U.S.Environmental Protection Agency, 1986a). Estimated median global atmospheric emission for anthropogenic and natural sources are 332×10^6 kg/year and 12×10^6 kg/year, respectively (summarized by Giusti et al., 1993). Inorganic and metallic Pb compounds are nonvolatile and will partition to airborne particulates or water vapors (Syracuse Research Corporation., 1999). Dispersion and deposition of Pb is dependent on the particle size (U.S.Environmental Protection Agency, 1986a; Syracuse Research Corporation., 1999). More soluble forms of Pb will be removed from the atmosphere by washout in rain.

In addition to atmospheric deposition, Pb may enter aquatic ecosystems through industrial or municipal wastewater effluents, storm water runoff, erosion, or direct point source inputs (e.g., Pb shot or accidental spills). Once in the aquatic environment, Pb will partition between the various compartments of the system (e.g., dissolved phase, solid phase, biota). The movement of Pb between dissolved and particulate forms is governed by factors such as pH, sorption, and biotransformation (see Section 8.2.3.2). Lead bound to organic matter will settle to the bottom sediment layer, be assimilated by aquatic organisms, or be resuspended in the water column. The uptake, accumulation, and toxicity of Pb in aquatic organisms from water and sediments are influenced by various environmental factors (e.g., pH, organic matter, temperature, hardness, bioavailability). These factors are further described in Sections 8.2.4.5.3 and 8.2.4.5.4). The remainder of this section discusses some methods for describing the distribution of atmospheric Pb in the aquatic environment.

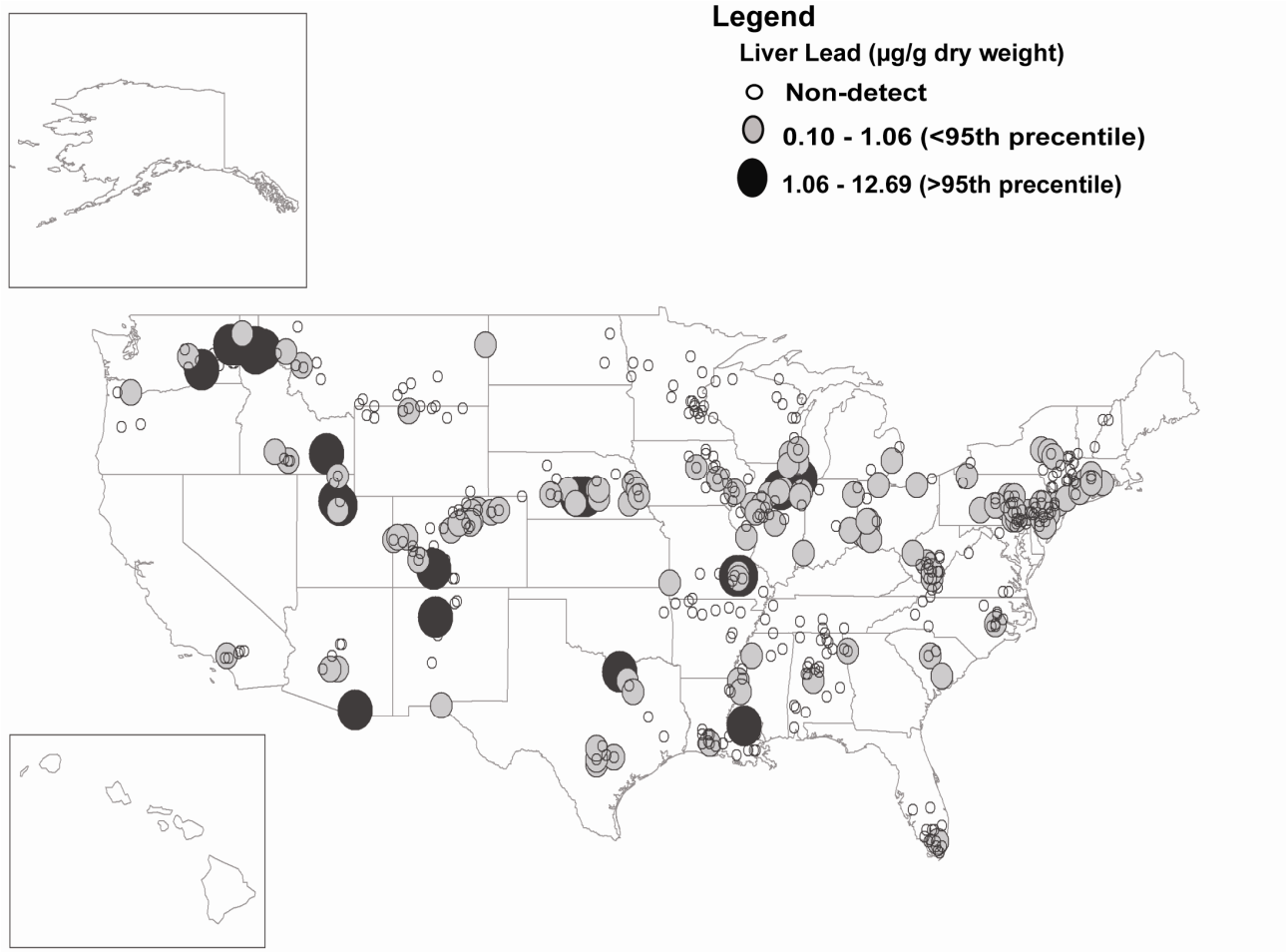


Figure 8-2.3.12. Spatial distribution of lead in liver tissues (N = 559).

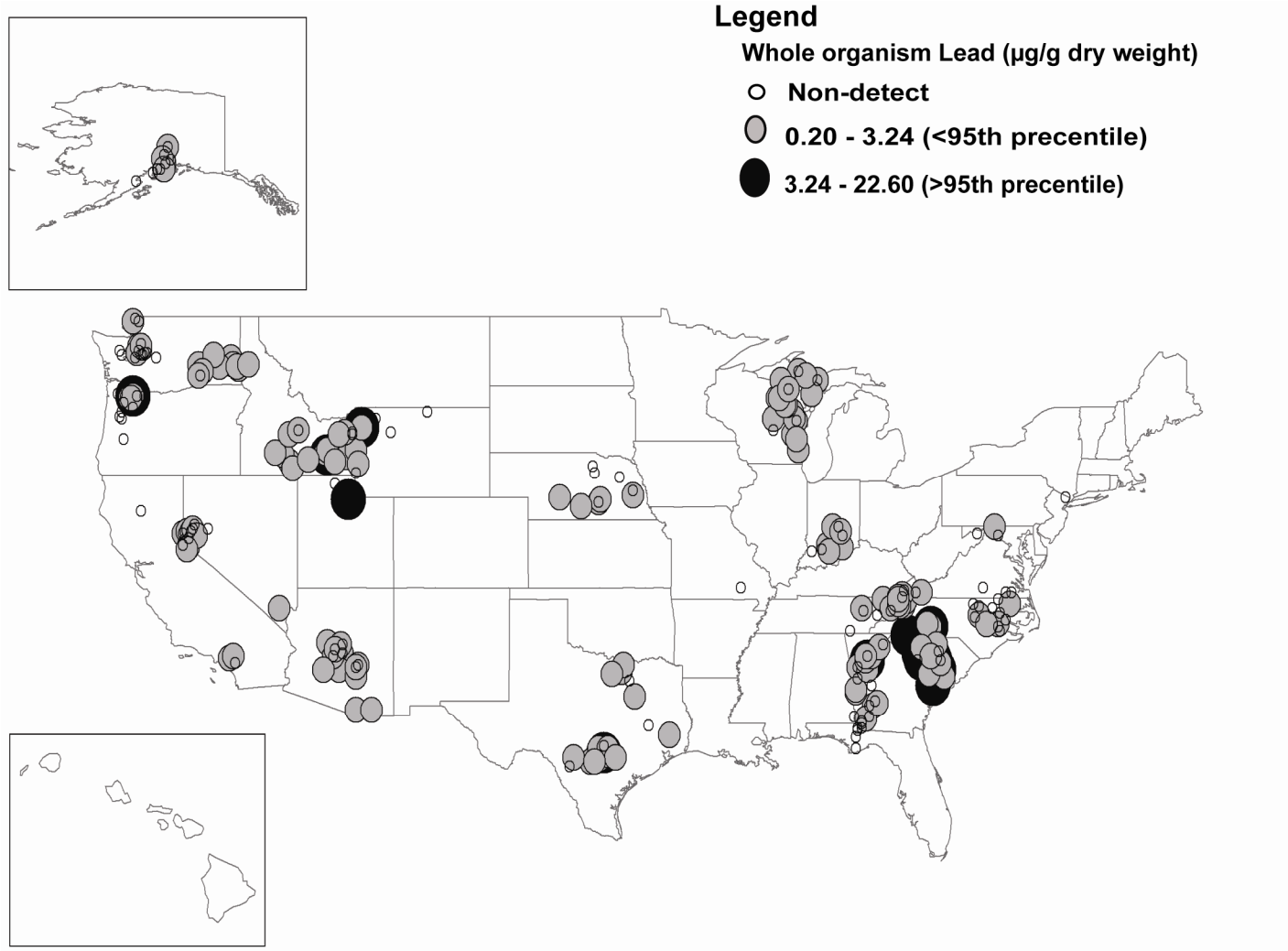


Figure 8-2.3.13. Spatial distribution of lead in whole organism tissues (N = 332).

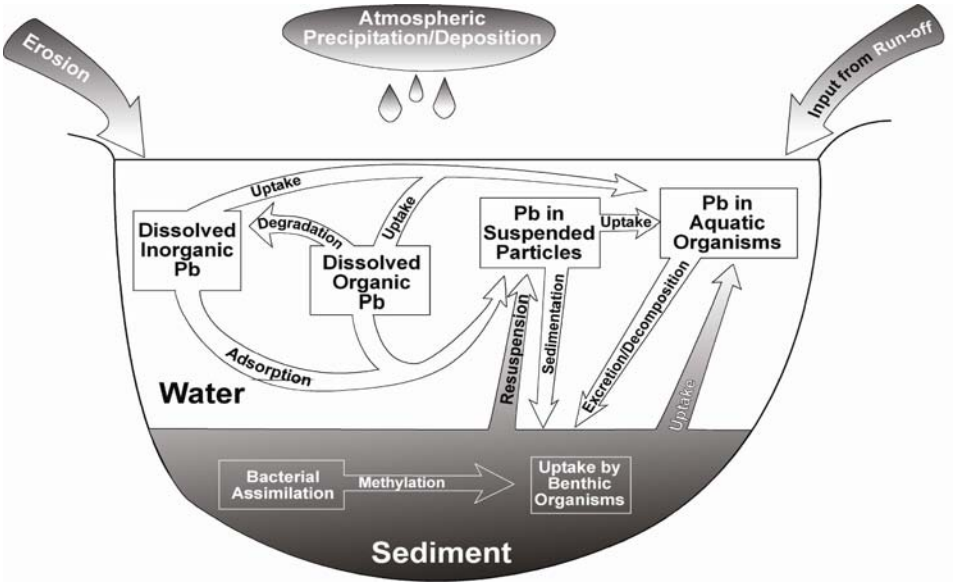


Figure 8-2.3.14. Lead cycle in an aquatic ecosystem.

Sediment Core Dating and Source Tracing

In addition to directly measuring Pb concentrations in various aquatic compartments (see Section 8.2.3.3), it is useful to study the vertical distribution of Pb. Sediment profiling and core dating is a method used to determine the extent of accumulation of atmospheric Pb and provide information on potential anthropogenic sources. Sediment concentration profiles are typically coupled with Pb isotopic analysis. The isotope fingerprinting method utilizes measurements of the abundance of common Pb isotopes (i.e., ^{204}Pb, ^{206}Pb, ^{207}Pb, ^{208}Pb) to distinguish between natural Pb over geologic time and potential anthropogenic sources. Details of this method were described in Section 8.1.2. The concentration of isotope ^{204}Pb has remained constant throughout time, while the other isotope species can be linked to various anthropogenic Pb sources. Typically, the ratios or signatures of isotopes (e.g., ^{207}Pb:^{206}Pb) are compared between environmental samples to indicate similarities or differences in the site being investigated and the potential known sources.

Generally, Pb concentrations in sediment vary with depth. For example, Chow et al. (1973) examined sediment Pb profiles in southern California. Lead concentrations were increased in the shallower sediment depths and comparatively decreased at greater

depths. These changes in sediment vertical concentration were attributed to higher anthropogenic Pb fluxes from municipal sewage, storm runoff, and atmospheric deposition. Similar experiments conducted throughout the United States have also suggested an increase in Pb concentrations in the upper sediment layer concomitant with increases in anthropogenic inputs (Bloom and Crecelius, 1987; Case et al., 1989; Ritson et al., 1999; Chillrud et al., 2003).

Sediment Pb concentration profiles and isotope analysis have also been used to identify specific anthropogenic sources. For example, Flegal et al. (1987) used isotopic ratios to trace sources of Pb in mussels from Monterey Bay, CA to a specific slag deposit. Several investigators have examined isotopic tracers to determine potential regional sources of Pb in eastern North America and the Great Lakes (Flegal et al., 1989b; Graney et al., 1995; Blais, 1996). Water samples from Lake Erie and Lake Ontario were collected and analyzed. Lead isotope ratios (^{206}Pb:^{207}Pb) from the lakes were compared to known ratios for Pb aerosols derived from industrial sources in Canada and the United States and found to correlate positively. This indicated that a majority of Pb in the lakes was derived from those industrial sources (Flegal et al., 1989b). Lead isotopes in sediment cores from Quebec and Ontario, Canada were also used to distinguish between the amount of Pb deposited from local Canadian sources (28.4 to 61.7%) and U.S. sources (38.3 to 71.6%) (Blais, 1996). Examination of Pb isotopes in sediment and suspended sediment in the St. Lawrence River were used to identify potential anthropogenic Pb sources from Canada (Gobeil et al., 1995, 2005). Graney et al. (1995) used Pb isotope measurements to describe the differing historic sources of Pb in Lake Erie, Ontario and in Michigan. Temporal changes in Pb isotopic ratios were found to correspond to sources such as regional deforestation from 1860 through1890, coal combustion and or smelting through 1930, and the influence of leaded gasoline consumption from 1930 to 1980.

The historic record of atmospheric Pb pollution has been studied to understand the natural background Pb concentration and the effects of Pb accumulation on ecosystems (Bindler et al., 1999; Renberg et al., 2000, 2002; Brännvall et al., 2001a,b). The most extensive work in this area has been conducted at pristine locations in Sweden (Bindler et al., 1999). In this study, soil, sediment, and tree rings were sampled for Pb

concentrations and isotopic analyses were conducted on the soil samples. From this record, historic Pb concentrations and Pb accumulation rates were estimated. Present day concentrations in the forest soils ranged from 40 to 100 mg/kg, while a natural background concentration was estimated at <1 mg/kg. The authors were able to model Pb concentrations for the past 6,000 years and also to project Pb concentrations for the next 400 years, given an assumed atmospheric deposition rate of 1 mg Pb m^{-2}/year. Models such as this are useful tools in determining the critical limits of metals in soils or sediments (Bindler et al., 1999; Renberg et al., 2002).

Lead source association may also be assessed through retrospective measurements. Squire et al. (2002) used a time-series approach to evaluate the change in Pb in San Francisco Bay, CA from 1989 to 1999. This approach involved the use of detailed linear regression models and long-term monitoring data to determine changes in Pb concentrations and to identify events corresponding to those changes. Sediment and water samples were collected throughout the bay and combined with data on effluent discharges, urban runoff, atmospheric deposition, and river discharges. The authors identified a 40% decline of Pb in the southern portion of the bay but found no change in the northern reach. The decline was attributed to a reduction in wastewater source loadings over the previous decade.

8.2.3.5 Summary

Lead is widely distributed in aquatic ecosystems, predominantly originating from atmospheric deposition or point source contribution. The fate and behavior of Pb in aquatic systems is regulated by physical and chemical factors such as pH, salinity, sediment sorption, transformation, and uptake by aquatic biota. In the United States, Pb concentrations in surface waters, sediments, and fish tissues range from 0.04 to 30 μg/L, 0.5 to 12,000 mg/kg, and 0.08 to 23 mg/kg, respectively. Atmospheric sources are generally decreasing, as the United States has removed Pb from gasoline and other products. However, elevated Pb concentrations remain at sites associated with mining wastes or wastewater effluents. Since the 1986 Pb AQCD, much has been learned about the processes affecting Pb fate and transport. Detailed analyses are currently available (i.e., Pb isotope dating) to allow for constructing the history of Pb accumulation and

identifying specific Pb contaminant sources. Continued source control along with examination of the physical and chemical properties will further allow for the reduction of Pb concentrations throughout the United States.

8.2.4 Species Response/Mode of Action

8.2.4.1 Introduction

Recent advancements in understanding the responses of aquatic biota to Pb exposure are highlighted in this section. A summary of the conclusions on the review of aquatic responses to Pb from the appropriate sections of the 1986 Pb AQCD, Volume II (U.S. Environmental Protection Agency, 1986a) and the subsequent conclusions and recommendations contained in the EPA staff review of that document (U.S. Environmental Protection Agency, 1990) are also provided. In addition, this section summarizes research subsequent to the 1986 Pb AQCD on Pb uptake into aquatic biota, effects of Pb speciation on uptake, resistance mechanisms to Pb toxicity, physiological effects of Pb, factors that affect responses to Pb, and factors associated with global climate change. Areas of research that are not addressed here include literature related to exposure to Pb shot or pellets and studies that examine human health-related endpoints (e.g., hypertension), which are described in other sections of this document.

8.2.4.2 Lead Uptake

Lead is nutritionally nonessential and non-beneficial and is toxic to living organisms in all of its forms (Eisler, 2000). Lead can bioaccumulate in the tissues of aquatic organisms through ingestion of food and water and adsorption from water (Vázquez et al., 1999; Vink, 2002) and subsequently lead to adverse effects (see Section 8.2.5). Lead concentrations in the tissues of aquatic organisms are generally higher in algae and benthic organisms and lower in higher trophic-level consumers (Eisler, 2000). Metals are not metabolized; therefore, they are good integrative indicators of exposure in aquatic biota (Luoma and Rainbow, 2005). Metal uptake is complex, being influenced by geochemistry, route of exposure (diet and adsorption), depuration, and growth (Luoma and Rainbow, 2005). This section discusses the factors affecting uptake of Pb by aquatic biota and the state of current research in this area.

As described in Section 8.2.3, the solubility of Pb in water varies with pH, temperature, and ion concentration (water hardness) (Weber, 1993). Lead becomes soluble and bioavailable under conditions of low pH, organic carbon content, suspended sediment concentrations, and ionic concentrations (i.e., low Cd, Ca, Fe, Mn, Zn) (Eisler, 2000). Lead rapidly loses solubility above pH 6.5 (Rickard and Nriagu, 1978) and precipitates out as $Pb(OH)^+$ and $PbHCO_3^+$ into bed sediments. However, at reduced pH levels or ionic concentrations, sediment Pb can remobilize and potentially become bioavailable to aquatic organisms (Weber, 1993).

The most bioavailable inorganic form of Pb is divalent Pb (Pb^{2+}), which tends to be more readily assimilated by organisms than complexed forms (Erten-Unal et al., 1998). On the other hand, the low solubility of Pb salts restricts movement across cell membranes, resulting in less accumulation of Pb in fish in comparison to other metals (e.g., Hg, Cu) (Baatrup, 1991).

The accumulation of Pb in aquatic organisms is, therefore, influenced by water pH, with lower pHs favoring bioavailability and accumulation. For example, fish accumulated Pb at a greater rate in acidic lakes (pH = 4.9 to 5.4) than in more neutral lakes (pH = 5.8 to 6.8) (Stripp et al., 1990). Merlini and Pozzi (1977) found that pumpkinseed sunfish exposed to Pb at pH 6.0 accumulated three-times as much Pb as fish kept at pH 7.5. However, Albers and Camardese (1993a,b) examined the effects of pH on Pb uptake in aquatic plants and invertebrates in acidic (pH ~5.0) and nonacidic (pH ~6.5) constructed wetlands, ponds, and small lakes in Maine and Maryland. Their results suggested that low pH had little effect on the accumulation of metals by aquatic plants and insects and on the concentration of metals in the waters of these aquatic systems (Albers and Camardese, 1993a,b).

Three geochemical factors that influence metal bioaccumulation in aquatic organisms include speciation, particulate metal form, and metal form in the tissues of prey items (Luoma and Rainbow, 2005). Lead is typically present in acidic aquatic environments as $PbSO_4$, $PbCl_4$, ionic Pb, cationic forms of Pb-hydroxide, and ordinary hydroxide $Pb(OH)_2$. In alkaline waters, common species of Pb include anionic forms of Pb-carbonate ($Pb(CO_3)$) and $Pb(OH)_2$. Labile forms of Pb (e.g., Pb^{2+}, $PbOH^+$, $PbCO_3$) are a significant portion of the Pb inputs to aquatic systems from atmospheric washout.

Particulate-bound forms are more often linked to urban runoff and mining effluents (Eisler, 2000). Little research has been done to link the complex concepts of chemical speciation and bioavailability in natural systems (Vink, 2002). The relationship between the geochemistry of the underlying sediment and the impact of temporal changes (e.g., seasonal temperatures) to metal speciation are particularly not well studied (Vink, 2002; Hassler et al., 2004).

Generally speaking, aquatic organisms exhibit three Pb accumulation strategies: (1) accumulation of significant Pb concentrations with a low rate of loss, (2) excretion of Pb roughly in balance with availability of metal in the environment, and (3) weak net accumulation due to very low metal uptake rate and no significant excretion (Rainbow, 1996). Species that accumulate nonessential metals such as Pb and that have low rates of loss must partition it internally in such a way that it is sparingly available metabolically. Otherwise, it may cause adverse toxicological effects (Rainbow, 1996). Aquatic organisms that exhibit this type of physiological response have been recommended for use both as environmental indicators of heavy metal pollution (Borgmann et al., 1993; Castro et al., 1996; Carter and Porter, 1997) and, in the case of macrophytes, as phytoremediators, because they accumulate heavy metals rapidly from surface water and sediment (Gavrilenko and Zolotukhina, 1989; Simòes Gonçalves et al., 1991; Carter and Porter, 1997).

Uptake experiments with aquatic plants and invertebrates (e.g., macrophytes, chironomids, crayfish) have shown steady increases in Pb uptake with increasing Pb concentration in solution (Knowlton et al., 1983; Timmermans et al., 1992). In crayfish, the process of molting can cause a reduction in body Pb concentrations, as Pb incorporated into the crayfish shell is eliminated (Knowlton et al., 1983). Vázquez et al. (1999) reported on the uptake of Pb from solution to the extracellular and intracellular compartments of 3 species of aquatic bryophytes. Relative to the 6 metals tested, Pb was found to accumulate to the largest degree in the extracellular compartments of all 3 bryophytes. The extracellular metals were defined as those that are incorporated into the cell wall or are found on the outer surface of the plasma membrane (i.e., adsorbed) (Vázquez et al., 1999). Intracellular metals were defined as metals introduced into the cell through a metabolically controlled process.

Arai et al. (2002) examined the effect of growth on the uptake and elimination of trace metals in the abalone *Haliotos*. They reported that older abalones had generally lower whole body concentrations of heavy metals than did younger, rapidly growing individuals. During the rapid growth of juveniles, the organism is less able to distinguish between essential (e.g., Zn), and nonessential metals (e.g., Pb). Once they reach maturity, they develop the ability to differentiate these metals. Li et al. (2004) reported a similar response in zebra fish embryo-larvae. Li et al. (2004) suggested that mature physiological systems are not developed in the embryo-larvae to handle elevated concentrations of metals. Therefore, metals are transported into the body by facilitated diffusion. Both the zebra fish and juvenile abalone demonstrate a rapid accumulation strategy followed by a low rate of loss as described above. There are insufficient data available to determine whether this phenomenon is true for other aquatic organisms. Growth rates are generally thought to be an important consideration in the comparison of Pb levels in individuals of the same species. The larger the individual the more the metal content is diluted by body tissue (Rainbow, 1996).

Once Pb is absorbed it may sequester into varying parts of the organism. Calcium appears to have an important influence on Pb transfer. For example, Pb uptake and retention in the skin and skeleton of coho salmon was reduced when dietary Ca was increased (Varanasi and Gmur, 1978). Organic Pb compounds tend to accumulate in lipids, and are taken up and accumulated in fish more readily than inorganic Pb compounds (Pattee and Pain, 2003).

Given the complexities of metal uptake in natural systems, a model incorporating some of the factors mentioned above is desirable. The EPA Environmental Research Laboratory developed a thermodynamic equilibrium model, MINTEQ that predicts aqueous speciation, adsorption, precipitation, and/or dissolution of solids for a defined set of environmental conditions (MacDonald et al., 2002; Playle, 2004). Although not specifically designed to model uptake, MINTEQ provides an indication of what forms of the metal are likely to be encountered by aquatic organisms by estimating the formation of metal ions, complexation of metals, and the general bioavailability of metals from environmental parameters. More recently, a mechanistic model centered on biodynamics has been proposed by Luoma and Rainbow (2005) as a method of tying together

geochemical influences, biological differences, and differences among metals to model metal bioaccumulation. The biodynamic model would be useful in determining the potential adverse effects on aquatic biota, which species are most useful as indicators of metal effects, and how ecosystems may change when contaminated by metals.

Two prominent models examine trace metal bioavailability and its link to effects (Hassler et al., 2004). These include the free ion activity model (FIAM) and the biotic ligand model (BLM), which is partially based upon FIAM. Generally, FIAM explores the activity of free ions in solution. More specific information on these models is provided in Section 8.2.2. The FIAM has been used to examine cationic binding to sensitive sites in algae and takes into account dissolved organic matter in complexation reactions (Niyogi and Wood, 2004). The BLM explores the activity of free ions at biologically reactive sites (i.e., fish gill tissue). Both of these models can increase our awareness of the processes governing the movement of Pb into aquatic biota. They provide insight into the speciation of Pb under certain environmental conditions (e.g., pH, DOC, hardness) and are important in helping understand how Pb and other metals move, accumulate, and cause effects in aquatic organisms. To date, there has been no BLM model of Pb, although research has been conducted on a Pb-gill binding model for rainbow trout (MacDonald et al., 2002; Niyogi and Wood, 2003, 2004).

Bioconcentration Factors (BCF)

BCFs for Pb are reported for various aquatic plants in Table 8-2.4.1. The green alga *Cladophora glomerata* is reported as having the highest BCF (Keeney et al., 1976). Duckweed (*Lemna minor*) exhibited high BCF values ranging from 840 to 3560 depending on the method of measurement (Rahmani and Sternberg, 1999). Duckweed that was either previously exposed or not exposed to Pb was exposed to a single dose of Pb-nitrate at 5000 µg/L for 21 days. Duckweed that was previously exposed to Pb removed 70 to 80% of the Pb from the water, while the previously unexposed duckweed removed 85 to 90%. Both plant groups were effective at removing Pb from the water at sublethal levels.

BCFs for Pb are reported for various invertebrates in Table 8-2.4.2. BCFs for freshwater snails were 738 for a 28-day exposure (Spehar et al., 1978) and 1,700 for a

Table 8-2.4.1. Bioconcentration Factors for Aquatic Plants

BCF	Species	Test Conditions	Reference
840 to 2700 (measured digestion)	Duckweed (*Lemna minor*)	21 days, Pb-nitrate	Rahmani and Sternberg (1999)
1150 to 3560 (measured solution)	Duckweed	21 days, Pb-nitrate	Rahmani and Sternberg (1999)
16,000 to 20,000	Alga *(Cladophora glomerata*)	not specified	Keeney et al. (1976)

Table 8-2.4.2. Bioconcentration Factors for Aquatic Invertebrates

BCF	Species	Test Conditions	Reference
738	Snail (*Physa integra*)	28 days, Pb-nitrate	Spehar et al. (1978)
1700	Snail (*Lymnaea palustris*)	120 days, Pb-nitrate	Borgmann et al. (1978)
499	Caddis fly (*Brachycentrus sp.*)	28 days, Pb-nitrate	Spehar et al. (1978)
1120	Stonefly (*Pteronarcys dorsata*)	28 days, Pb-nitrate	Spehar et al. (1978)
1930	Scud (*Hyalella azteca*)	4 days, Pb-chloride	MacLean et al. (1996)
3670	Midge larvae (*Chironomus riparius*)	28 days	Timmermans et al. (1992)

120-day exposure (Borgmann et al., 1978). Other reported values for invertebrates included a BCF of 1930 for the scud during a 4-day exposure (MacLean et al., 1996), and BCFs of 499 and 1120 for the caddis fly and stonefly, respectively, in 28-day exposures (Spehar et al., 1978). In a 28-day exposure, midge larvae were reported with a BCF of 3670 (Timmermans et al., 1992).

BCFs for freshwater fish were 42 and 45 for brook trout and bluegill, respectively (Holcombe et al., 1976; Atchison et al., 1977). Although no BCFs have been reported for amphibians, Pb-nitrate was reported to accumulate mainly in the ventral skin and in the kidneys of frogs (Vogiatzis and Loumbourdis, 1999).

8.2.4.3 Resistance Mechanisms

8.2.4.3.1 Detoxification Mechanisms

Detoxification includes the biological processes by which the toxic qualities, or the probability and/or severity of harmful effects, of a poison or toxin are reduced by the organism. In the case of heavy metals, this process frequently involves the sequestration of the metal, rendering it metabolically inactive. Recent research into heavy metal detoxification in aquatic biota has focused on several physiological and biochemical mechanisms for detoxifying Pb. This section examines these mechanisms and the ability of plants, protists, invertebrates, and fish to mitigate Pb toxicity.

Plants and Protists

Deng et al. (2004) studied the uptake and translocation of Pb in wetland plant species surviving in contaminated sites. They found that all plants tended to sequester significantly larger amounts of Pb in their roots than in their shoots. Deng et al. (2004) calculated a translocation factor (TF), the amount of Pb found in the shoots divided by the amount of Pb found in the root system, and found that TFs ranged from 0.02 to 0.80. Concentrations of Pb in shoots were maintained at low levels and varied within a narrow range. Deng et al. (2004) observed that plants grown in Pb-contaminated sites usually contained higher concentrations than the 27 mg/kg toxicity threshold established for plants by Beckett and Davis (1977). Some of the wetland plants examined by Deng et al. (2004) also accumulated high concentrations of metals in shoot tissues; however, these metals were assumed to be detoxified (metabolically unavailable), as no toxic response to these elevated concentrations was observed. Deng et al. (2004) suggested that this ability is likely related to discrete internal metal detoxification tolerance mechanisms.

Phytochelatins are thiol-containing intracellular metal-binding polypeptides that are produced by plants and protists in response to excessive uptake of heavy metals (Zenk, 1996). Phytochelatins are synthesized by the enzyme phytochelatin synthase that is activated by the presence of metal ions and uses glutathione as a substrate. When phytochelatins are synthesized in sufficient amounts to chelate the metal ion, the enzyme is deactivated (Morelli and Scarano, 2001).

Morelli and Scarano (2001) studied phytochelatin synthesis and stability in the marine diatom *Phaeodactylum tricornutum* in the presence of Pb. They found that when metal exposure was alleviated, significant cellular Pb-phytochelatin complex content was lost. Their findings support a hypothesis of vacuolarization proposed for higher plants (Zenk, 1996), in which metal-phytochelatin complexes are actively transported from the cytosol to the vacuole, where they undergo rapid turnover. Zenk (1996) suggested that the complex dissociates, and the metal-free peptide is subsequently degraded. Morelli and Scarano (2001) proposed concomitant occurrence of phytochelatin synthesis and release during metal exposure, as a coincident detoxification mechanism in *P. tricornutum*.

Aquatic Invertebrates

Like plants and protists, aquatic animals detoxify Pb by preventing it from being metabolically available, though their mechanisms for doing so vary. Invertebrates use lysosomal-vacuolar systems to sequester and process Pb within glandular cells (Giamberini and Pihan, 1996). They also accumulate Pb as deposits on and within skeletal tissue (Knowlton et al., 1983; Anderson et al., 1997; Boisson et al., 2002), and some can efficiently excrete Pb (Vogt and Quinitio, 1994; Prasuna et al., 1996).

Boisson et al. (2002) used radiotracers to evaluate the transfer of Pb into the food pathway of the starfish *Asterias rubens* as well as its distribution and retention in various body compartments. Boisson et al. (2002) monitored Pb elimination after a single feeding of Pb-contaminated molluscs and found that Pb was sequestered and retained in the skeleton of the starfish, preventing it from being metabolically available in other tissues. Elimination (as percent retention in the skeleton) was found to follow an exponential time course. Elimination was rapid at first, but slowed after 1 week, and eventually stabilized, implying an infinite biological half-life for firmly bound Pb. Results of radiotracer tracking suggest that Pb migrates within the body wall from the organic matrix to the calcified skeleton. From there, the metal is either absorbed directly or adsorbed on newly-produced ossicles (small calcareous skeletal structures), where it is efficiently retained as mineral deposition and is not metabolically active (Boisson et al., 2002).

AbdAllah and Moustafa (2002) studied the Pb storage capability of organs in the marine snail *Nerita saxtilis*. Enlarged electron-dense vesicles and many granules were observed in digestive cells of these snails and are suggested to be the site of storage of detoxified metals. *N. saxtilis* were found to be capable of concentrating Pb up to 50 times that of surrounding marine water without exhibiting signs of histopathologic changes. This ability has been attributed to chelation with various biochemical compounds, such as thionine (forming metallothionine) (Rainbow, 1996), or complexation with carbonate, forming lipofuchsin (AbdAllah and Moustafa, 2002). Granules observed in lysosomal residual bodies were presumed to be the result of Pb accumulation. The presence of large vacuoles and residual bodies were indicative of the fragmentation phase of digestion, suggesting that Pb was also processed chemically in the digestive cells.

The podocyte cells of the pericardial gland of bivalves are involved in the ultrafiltration of the hemolymph (Giamberini and Pihan, 1996). A microanalytical study of the podocytes in *Dreissena polymorpha* exposed to Pb revealed lysosomal-vacuolar storage/processing similar to that in the digestive cells of *Nerita saxtilis.* The lysosome is thought to be the target organelle for trace metal accumulation in various organs of bivalves (Giamberini and Pihan, 1996). Epithelial secretion is the principal detoxification mechanism of the tiger prawn *Penaeus monodon*. Vogt and Quinitio (1994) found that Pb granules tended to accumulate in the epithelial cells of the antennal gland (the organ of excretion) of juveniles exposed for 5 and 10 days to waterborne Pb. The metal is deposited in vacuoles belonging to the lysosomal system. Continued deposition leads to the formation of electron-dense granules. Mature granules are released from the cells by apocrine secretion into the lumen of the gland, and presumably excreted through the nephridopore (i.e., the opening of the antennal gland). Apocrine secretion is predominant, so that as granules form, they are kept at low levels. Excretion was also found to be a primary and efficient detoxification mechanism in the shrimp *Chrissia halyi* (Prasuna et al., 1996).

Crayfish exposed to Pb have been shown to concentrate the metal in their exoskeleton and exuvia through adsorption processes. More than 80% of Pb found in exposed crayfish has been found in exoskeletons (Knowlton et al., 1983; Anderson et al., 1997). Following exposure, clearance is most dramatic from the exoskeleton. The result

of a 3-week Pb-clearance study with red swamp crayfish *Procambarus clarkia*, following a 7-week exposure to 150 µg/L Pb, showed an 87% clearance from the exoskeleton due, in part, to molting. Other organs or tissues that take up significant amounts of Pb include the gills, hepatopancreas, muscle, and hemolymph, in decreasing order. These parts cleared >50% of accumulated Pb over the 3-week clearance period, with the exception of the hepatopancreas. The hepatopancreas is the organ of metal storage and detoxification, although the molecular mechanisms of metal balance in crayfish have yet to be extensively investigated (Anderson et al., 1997).

Fish

Most fish use mucus as a first line of defense against heavy metals (Coello and Khan, 1996). In fish, some epithelia are covered with extracellular mucus secreted from specialized cells. Mucus contains glycoproteins, and composition varies among species. Mucosal glycoproteins chelate Pb, and settle, removing the metal from the water column. Fish may secrete large amounts of mucus when they come into contact with potential chemical and biochemical threats. Coello and Khan (1996) investigated the role of externally added fish mucus and scales in accumulating Pb from water, and the relationship of these with the toxicity of Pb in fingerlings of green sunfish, goldfish and largemouth bass. The authors compared trials in which fish scales from black sea bass (*Centropristis striata*) and flounder (*Pseudopleuronectes americanus*) and mucus from largemouth bass were added to green sunfish, goldfish, and largemouth bass test systems and to reference test systems. On exposure to Pb, fish immediately started secreting mucus from epidermal cells in various parts of the body. Metallic Pb stimulated filamentous secretion, mostly from the ventrolateral areas of the gills, while Pb-nitrate stimulated diffuse molecular mucus secretion from all over the body. The addition of largemouth bass mucus significantly increased the LT_{50} (the time to kill 50%) for green sunfish and goldfish exposed to 250 mg/L of Pb-nitrate. In contrast, Tao et al. (2000) found that mucus reduced the overall bioavailability of Pb to fish but that the reduction was insignificant. Coello and Khan (1996) found that scales were more significant in reducing LT_{50} than mucous. Fish scales can accumulate high concentrations of metals, including Pb, through chelation with keratin. Scales were shown to buffer the pH of

Pb-nitrate in solution and remove Pb from water after which they settled out of the water column. Addition of scales to test water made all species (green sunfish, goldfish, and largemouth bass) more tolerant of Pb.

Summary of Detoxifiction Processes

Mechanisms of detoxification vary among aquatic biota and include processes such as translocation, excretion, chelation, adsorption, vacuolar storage, and deposition. Lead detoxification has not been studied extensively in aquatic organisms, but existing results indicate the following:

- Protists and plants produce intracellular polypeptides that form complexes with Pb (Zenk, 1996; Morelli and Scarano, 2001).
- Macrophytes and wetland plants that thrive in Pb-contaminated regions have developed translocation strategies for tolerance and detoxification (Knowlton et al., 1983; Deng et al., 2004).
- Some starfish (asteroids) sequester the metal via mineral deposition into the exoskeleton (Boisson et al., 2002).
- Species of mollusc employ lysosomal-vacuolar systems that store and chemically process Pb in the cells of their digestive and pericardial glands (Giamberini and Pihan, 1996; AbdAllah and Moustafa, 2002).
- Decapods can efficiently excrete Pb (Vogt and Quinitio, 1994; Giamberini and Pihan, 1996) and sequester metal through adsorption to the exoskeleton (Knowlton et al., 1983).
- Fish scales and mucous chelate Pb in the water column, and potentially reduce visceral exposure.

8.2.4.3.2 Avoidance Response

Avoidance is the evasion of a perceived threat. Recent research into heavy metal avoidance in aquatic organisms has looked at dose-response relationships as well as the effects of coincident environmental factors. Preference/avoidance response to Pb has not been extensively studied in aquatic organisms. In particular, data for aquatic invertebrates is lacking. Using recent literature, this section examines preference-

avoidance responses of invertebrates and fish to Pb and some other environmental gradients.

Aquatic Invertebrates

Only one study was identified on avoidance response in aquatic invertebrates. Lefcort et al. (2004) studied the avoidance behavior of the aquatic pulmonate snail *Physella columbiana* from a pond that had been polluted with heavy metals for over 120 years. In a Y-maze test, first generation *P. columbiana* from the contaminated site avoided Pb at 9283 µg/L ($p < 0.05$) and moved toward Pb at 6255 µg/L ($p < 0.05$). It is thought that attraction to Pb at certain elevated concentrations is related to Pb neuron-stimulating properties (Lefcort et al., 2004). These results are consistent with those from similar studies. Control snails from reference sites, and first and second generation snails from contaminated sites were capable of detecting and avoiding heavy metals, although the first generation was better than the second generation, and the second was better than the controls at doing so. This suggests that detection and avoidance of Pb is both genetic and environmentally based for *P. columbiana.* Lefcort et al. (2004) observed heightened sensitivity to, and avoidance of, heavy metals by the snails when metals where present in combination.

Aquatic Vertebrates

Steele et al. (1989) studied the preference-avoidance response of bullfrog (*Rana catesbeiana*) to plumes of Pb-contaminated water following 144 h exposure to 0 to 1000 µg Pb/L. In this laboratory experiment, tadpoles were exposed to an influx of 1000 µg/L at five different infusion rates (i.e., volumes per unit time into the test system). Experiments were videotaped and location data from the tank were used to assess response. No significant differences were seen in preference-avoidance responses to Pb either nonexposed or previously exposed animals. In a similar subsequent study, Steele et al. (1991) studied preference-avoidance response to Pb in American toad (*Bufo mericanus*) using the same exposure range (0 to 1000 µg/L). *B. americanus* did not significantly avoid Pb, and behavioral stress responses were not observed. The results do not indicate whether the tadpoles were capable of perceiving the contaminant. Lack of

avoidance may indicate insufficient perception or the lack of physiological stress (Steele et al., 1991).

The olfactory system in fish is involved in their forming avoidance response to heavy metals (Brown et al., 1982; Svecevičius, 1991). It is generally thought that behavioral avoidance of contaminants may be a cause of reduced fish populations in some water bodies, because of disturbances in migration and distribution patterns (Svecevičius, 2001). Unfortunately, avoidance of Pb by fish has not been studied as extensively as for other heavy metals (Woodward et al., 1995).

Woodward et al. (1995) studied metal mixture avoidance response in brown trout (*Salmo trutta*), as well as the added effects of acidification. A 1-fold (1×) mixture contained 1.1 µg/L Cd, 12 µg/L Cu, 55 µg/L Zn, and 3.2 µg/L Pb (all metals were in the form of chlorides). Avoidance was quantified as time spent in test water, trip time to test water, and number of trips. Brown trout avoided the 1× mixture as well as the 0.5×, 2×, 4×, and 10× mixtures, but not the 0.1× mixture. Reduced avoidance was observed at higher concentrations (4× and 10×). The authors proposed that the reduced avoidance response was due to impaired perception due to injury. These responses are typical of other fish species to individual metals of similar concentrations (Woodward et al., 1995). This study does not conclusively indicate which of the metals in the mixture may be causing the avoidance response. However, given the neurotoxic effects of Pb, impaired perception is a likely response of Pb-exposed fish.

When test water was reduced in pH from 8 to 7, 6 to 5, brown trout avoidance increased, but with no significant difference between metal treatments and controls. However, in the 1× metal mixture treatment, brown trout made fewer trips into the test water chamber at the lower pHs (Woodward et al., 1995). This response may be related to an increased abundance of Pb cations at lower pH values in the test system.

Scherer and McNicol (1998) investigated the preference-avoidance responses of lake whitefish (*Coregonus clupeaformis*) to overlapping gradients of light and Pb. Whitefish were found to prefer shade in untreated water. Lead concentrations under illumination ranged from 0 to 1000 µg/L, and from 0 to 54,000 µg/L in the shade. Under uniform illumination, Pb was avoided at concentrations above 10 µg/L, but avoidance behavior lacked a dose-dependent increase over concentrations ranging from 10 to

1000 µg/L Pb. Avoidance in shaded areas was strongly suppressed, and whitefish only avoided Pb at concentrations at or above 32,000 µg/L.

In summary, of those aquatic organisms studied, some are quite adept at avoiding Pb in aquatic systems, while others seem incapable of detecting its presence. Snails have been shown to be sensitive to Pb and to avoid it at high concentrations. Conversely, anuran (frog and toad) species lack an avoidance response to the metal. Fish avoidance of many chemical toxicants has been well established, and it is a dominant sublethal response in polluted waters (Svecevičius, 2001). However, no studies have been located specifically examining avoidance behavior for Pb in fish. Environmental gradients, such as light and pH, can alter preference-avoidance responses.

8.2.4.4 Physiological Effects of Lead

This section presents a review of the physiological effects and functional growth responses associated with the exposure of aquatic biota to Pb. Physiological effects of Pb on aquatic biota can occur at the biochemical, cellular, and tissue levels of organization and include inhibition of heme formation, adverse effects to blood chemistry, and decreases in enzyme levels. Functional growth responses resulting from Pb exposure include changes in growth patterns, gill binding affinities, and absorption rates.

Biochemical Effects

Lead was observed to have a gender-selective effect on brain endocannabinoid (eCB) (e.g., 2-arachidonylglycerol [2-AG] and *N*-arachidonylethanolamine [AEA]) levels in fathead minnow *Pimephales promelas* (Rademacher et al., 2005). Cannabinoids, such as eCB, influence locomotor activity in organisms. Increased levels of cannabinoids have been shown to stimulate locomotor activity and decreased levels slow locomotor activity (Sañudo-Peña et al., 2000). Male and female fathead minnows were exposed to 0 and 1000 µg/L of Pb. Female minnows in the control group contained significantly higher levels of AEA and 2-AG compared to males. At a concentration of 1000 µg/L, this pattern reversed, with males showing significantly higher levels of AEA in the brain than females (Rademacher et al., 2005). After 14-days exposure to the 1000 µg/L treatment,

significantly higher levels of 2-AG were found in male fathead minnows, but no effect on 2-AG levels in females was observed (Rademacher et al., 2005).

Lead acetate slightly inhibited 7-ethoxyresorufin-*o*-deethylase (7-EROD) activity in *Gammarus pulex* exposed for up to 96 h to a single toxicant concentration (EC_{50}) (Kutlu and Susuz, 2004). The exact concentration used in the study was not reported. The EROD enzyme is required to catalyze the conjugation and detoxification of toxic molecules and has been proposed as a biomarker for contaminant exposure. The authors believe more detailed studies are required to confirm EROD as a biomarker for Pb exposure. The enzyme group alanine transferases (ALT) has been suggested as a bioindicator/biomarker of Pb stress (Blasco and Puppo, 1999). A negative correlation was observed between Pb accumulation and ALT concentrations in the gills and soft body of *Ruditapes philippinarum* exposed to 350 to 700 µg/L of Pb for 7 days (Blasco and Puppo, 1999).

Studies have identified ALAD in fish and amphibians as a useful indicator of Pb exposure (Gill et al., 1991; Nakagawa et al., 1995a,b). ALAD catalyzes the formation of hemoglobin and early steps in the synthesis of protoporphyrin (Gill et al., 1991; Nakagawa et al., 1995b). The absence of an inhibitory effect on this enzyme following exposure to cadmium, copper, zinc, and mercury suggests that this enzyme reacts specifically to Pb (Johansson-Sjöbeck and Larsson, 1979; Gill et al., 1991). A 0% decrease in ALAD activity was reported in common carp (*Cyprinus carpio*) exposed to a Pb concentration of 10 µg/L for 20 days (Nakagawa et al., 1995b). The recovery the ALAD activity after exposure to Pb has also been examined in carp (Nakagawa et al., 1995a). After 2-week exposure to 200 µg/L, ALAD activity decreased to approximately 25% of value reported for controls (Nakagawa et al., 1995a). Fish removed from the test concentration after 2 weeks and placed in a Pb-free environment recovered slightly, but ALAD activity was only 50% of the controls even after 4 weeks (Nakagawa et al., 1995a). Vogiatzis and Loumbourdis (1999) exposed the frog (*Rana ridibunda*) to a Pb concentration of 14,000 µg/L over 30 days and a 90% decrease in ALAD activity was observed in the frogs.

Blood Chemistry

Numerous studies have examined the effects of Pb exposure on blood chemistry in aquatic biota. These studies have primarily used fish in acute and chronic exposures to Pb concentrations ranging from 100 to 10,000 µg/L. Decreased erythrocyte, hemoglobin, and hemocrit levels were observed in rosy barb (*Barbus puntius*) during an 8-week exposure to 126 µg/L of Pb-nitrate (Gill et al., 1991).

No difference was found in red blood cell counts and blood hemoglobin in yellow eels (*Anguilla anguilla*) exposed to 0 and 300 µg/L of Pb for 30 days (Santos and Hall, 1990). The number of white blood cells, in the form of lymphocytes, increased in the exposed eels. The authors concluded this demonstrates the lasting action of Pb as a toxicant on the immune system (Santos and Hall, 1990). Significant decreases in red blood cell counts and volume was reported in blue tilapia (*Oreochromis aureus*) exposed to Pb-chloride at a concentration of 10,000 µg/L for 1 week (Allen, 1993).

Blood components, such as plasma glucose, total plasma protein, and total plasma cholesterol, were unaffected in yellow eels exposed to 300 µg/L of Pb for 30 days (Santos and Hall, 1990). Effects on plasma chemistry were observed in *Oreochromis mossambicus* exposed to 0, 18,000, 24,000, and 33,000 µg/L of Pb (Ruparelia et al., 1989). Significant decreases in plasma glucose (hypoglycemic levels) were reported at concentrations of 24,000 and 33,000 μg/L after 14 and 21 days of exposure, and at 18,000 µg/L after 21 days of exposure (Ruparelia et al., 1989). Plasma cholesterol levels dropped significantly in comparison to controls after 14 days of exposure to 33,000 µg/L and in all test concentrations after 21 days of exposure (Ruparelia et al., 1989). Similarly, concentrations of blood serum protein, albumin, and globulin were identified as bioindicators of Pb stress in carp (*Cyprinus carpio*) exposed to Pb-nitrates at concentrations of 800 and 8000 µg/L (Gopal et al., 1997).

Tissues

In fish, the gills serve as an active site for ion uptake. Recent studies have examined the competition between cations for binding sites at the fish gill (e.g., Ca^{2+}, Mg^{2+}, Na^{+}, H^{+}, Pb^{2+}) (MacDonald et al., 2002; Rogers and Wood, 2003, 2004). Studies suggest that Pb^{2+} is an antagonist of Ca^{2+} uptake (Rogers and Wood, 2003, 2004).

MacDonald et al. (2002) proposed a gill-Pb binding model that assumes Pb^{2+} has a ≥100 times greater affinity for binding sites at the fish gill than other cations. More toxicity studies are required to quantify critical Pb burdens that could be used as indicators of Pb toxicity (Niyogi and Wood, 2003).

Growth Responses

A negative linear relationship was observed in the marine gastropod abalone (*Haliotis*) between shell length and muscle Pb concentrations (Arai et al., 2002). Abalones were collected from two sites along the Japanese coast. *Haliotis discus hannai* were collected from along the coast at Onagawa; *Haliotis discus* were collected from along the coast at Amatsu Kominato. The authors did not report significant differences between the two sampling sites. From samples collected at Onagawa, Pb concentrations of 0.03 and 0.01 μg/g were associated with abalone shell lengths of 7.7 cm (3 years old) and 12.3 cm (6 years old), respectively. From samples collected at Amatsu Kominato, Pb concentrations of 0.09 and 0.01 μg/g were associated with abalone shell lengths of 3.9 cm (0 years old) and 15.3 cm (8 years old), respectively (Arai et al., 2002). The authors theorized that young abalones, experiencing rapid growth, do not discriminate between the uptake of essential and nonessential metals. However, as abalones grow larger and their rate of growth decreases, they increasingly favor the uptake of essential metals over nonessential metals. This is demonstrated by the relatively consistent concentrations of Cu, Mn, and Zn that were reported for the abalone samples (Arai et al., 2002).

Other Physiological Effects

Increased levels of Pb in water were found to increase fish production of mucus: excess mucus coagulates were observed over the entire body of fishes. Buildup was particularly high around the gills, and in the worst cases, interfered with respiration and resulted in death by anoxia (Aronson, 1971; National Research Council of Canada., 1973).

8.2.4.5 Factors That Modify Organism Response to Lead

A great deal of research has been undertaken recently to better understand the factors that modify aquatic organism response to Pb. The driving force behind this research is the development of the BLM approach to AWQC development. A discussion of research on the many factors that can modify aquatic organism response to Pb is provided in this section.

8.2.4.5.1 Organism Age and Size Influence on Lead Uptake and Response

It is generally accepted that Pb accumulation in living organisms is controlled, in part, by metabolic rates (Farkas et al., 2003). Metabolic rates are, in-turn, controlled by the physiological conditions of an organism, including such factors as size, age, point in reproductive cycle, nutrition, and overall health. Of these physiological conditions, size and age are the most commonly investigated in relation to heavy metal uptake. This section reviews recent research focusing on relationships between body size, age, and Pb accumulation in aquatic invertebrates and fish.

Invertebrates

MacLean et al. (1996) investigated bioaccumulation kinetics and toxicity of Pb in the amphipod *Hyalella azteca*. Their results indicated that body size did not greatly influence Pb accumulation in *H. azteca* exposed to 50 or 100 µg/L of $PbCl_2$ for 4 days. Canli and Furness (1993) found similar results in the Norway lobster *Nephrops norvegicus* exposed to 100 µg/L of $Pb(NO_3)_2$ for 30 days. No significant sex- or size-related differences were found in concentrations of Pb in the tissue. The highest tissue burden was found in the carapaces (42%). Several studies have determined that Pb can bind to the exoskeleton of invertebrates and sometimes dominate the total Pb accumulated (Knowlton et al., 1983). This adsorption of Pb to the outer surface of invertebrates can result in strong negative relationships for whole-body Pb concentration as a function of body mass (i.e., concentrations decrease rapidly with increased body size and then stabilize) (MacLean et al., 1996).

Drava et al. (2004) investigated Pb concentrations in the muscle of red shrimp *Aristeus antennatus* from the northwest Mediterranean. Lead concentrations ranged from

0.04 to 0.31 µg/g dw. No significant relationships between size and Pb concentration in *A. antennatus* were found, and concentrations were not related to reproductive status.

Arai et al. (2002) analyzed abalones (*Haliotis*) at various life stages from coastal regions of Japan. They investigated growth effects on the uptake and elimination of Pb. Results indicated a significant negative linear relationship between age, shell length and Pb concentrations in muscle tissue. The relationship was consistent despite habitat variations in Pb concentrations between the study sites, suggesting that Pb concentrations changed with growth in the muscle tissue of test specimens and implying that abalone can mitigate Pb exposure as they age.

Fish

Douben (1989) investigated the effects of body size and age on Pb body burden in the stone loach (*Noemacheilus barbatulus* L.). Fish were caught during two consecutive springs from three Derbyshire rivers. Results indicated that Pb burden increased slightly with age. Similarly, Köck et al. (1996) found that concentrations of Pb in the liver and kidneys of Arctic char (*Salvelinus alpinus*) taken from oligotrophic alpine lakes were positively correlated with age. It has been suggested that fish are not able to eliminate Pb completely, and that this leads to a stepwise accumulation from year to year (Köck et al., 1996). In contrast, Farkas et al. (2003) found a negative relationship between Pb concentrations and muscle and gill Pb concentrations in the freshwater fish *Abramis brama*. Fish were taken from a low-contaminated site and contained between 0.44 and 3.24 µg/g Pb dw. Negative correlations between metal concentration and fish size in low-contaminated waters likely results from variations in feeding rates associated with developmental stages. This hypothesis is consistent with the fact that in low-contaminated waters, feeding is the main route of uptake and feeding rates decrease with development in fish (Farkas et al., 2003).

In summary, relationships between age, size, and Pb body burden in aquatic invertebrates and fish are interspecifically variable and depend on many environment-related variables (e.g., exposure) (Farkas et al., 2003).

8.2.4.5.2 Genetics

There are few studies documenting the effects of Pb on organismal and population genetics, although rapid advances in biotechnology have prompted recent research in this area (Beaty et al., 1998). There are two principal effects that sublethal exposure to a contaminant can have on the genetics of an organism and/or population: (1) a contaminant may influence selection by selecting for certain phenotypes that enable populations to better cope with the chemical; or (2) a contaminant can be genotoxic, meaning it can produce alterations in nucleic acids at sublethal exposure concentrations, resulting in changes in hereditary characteristics or DNA inactivation (Shugart, 1995). Laboratory studies have shown that exposure to Pb^{2+} at 10 mg/mL of blood produces chromosomal aberrations (i.e., deviations in the normal structure or number of chromosomes) in some organisms (Cestari et al., 2004). Effects of genotoxicity and toxin-induced selection do not preclude one another, and may act together on exposed populations. This section reviews Pb genotoxicity and the effects of Pb-induced selection in aquatic populations.

Selection

Evidence for genetic selection in the natural environment has been observed in some aquatic populations exposed to metals (Rand et al., 1995; Beaty et al., 1998; Duan et al., 2000; Kim et al., 2003). Because tolerant individuals have a selective advantage over vulnerable individuals in polluted environments, the frequency of tolerance genes will increase in exposed populations over time (Beaty et al., 1998). Several studies have shown that heavy metals can alter population gene pools in aquatic invertebrates. These changes have resulted in decreased genetic diversity and are thought to be a potential source of population instability (Duan et al., 2000; Kim et al., 2003).

Kim et al. (2003) investigated genetic differences and population structuring in the gastropod *Littorina brevicula* from heavy-metal polluted and unpolluted environments. Organisms from polluted sites contained a mean of 1.76 µg/g Pb, while organisms from unpolluted sites contained 0.33 µg/g Pb. They found significant differences in haplotypes between the test groups and allelic diversity was significantly lower among *L. brevicula* from polluted regions. In contrast, Yap et al. (2004) performed a similar

experiment with the green-lipped mussel *Perna viridis*; they found that mussels from contaminated sites containing between 4 and 10 µg/g Pb, as well as other heavy metals, exhibited a higher percentage of polymorphic loci and excess heterozygosity compared to those from uncontaminated sites. The higher level of genetic diversity was attributed to greater environmental heterogeneity (i.e., variation due to pollution gradients) in contaminated sites (Yap et al., 2004).

Duan et al. (2000) investigated amphipod (*Hyalella azteca*) selective mortality and genetic structure following acute exposure to Pb (5.47 mg/L $Pb(NO_2)_2$) as well as exposure to other heavy metals. They found that genetic differentiation consistently increased among survivors from the original population, supporting the hypothesis that heavy metals, including Pb, have the potential to alter the gene pools of aquatic organisms.

Genotoxicity

Low-level (50 µg/L) Pb exposure in water over 4 weeks resulted in DNA strand breakage in the freshwater mussel *Anodonta grandis* (Black et al., 1996), although higher concentrations (up to 5000 µg/L) did not result in significant breakage by the end of the study period. These results suggest that a threshold effect for DNA damage and repair exists, where DNA repair only occurs once a certain body exposure level has been reached. More recently, Cestari et al. (2004) observed similar results in neotropical fish (*Hoplias malabaricus*) that were fed Pb-contaminated food over 18, 41, and 64 days. Lead body burdens in *H. malabaricus* were approximately 21 µg Pb^{2+}/g. Results indicated that exposure to Pb significantly increased the frequency of chromosomal aberrations and DNA damage in kidney cell cultures, although when assessed at the end of the longer exposure periods, aberrations were less common.

8.2.4.5.3 Environmental Biological Factors

Environmental factors that are biological in origin can alter the availability, uptake and toxicity of Pb to aquatic organisms. These factors can be grouped into living and non-living constituents. For example, living organisms may sequester Pb from the water column, reducing the availability and toxicity of the metal in the water column to other

biota, thus reducing potential toxic effects in other organisms. Non-living organic material (e.g., components of sloughed-off scales, mucus, carcasses, and other decomposing, humic material) can similarly combine with Pb from the water column, rendering it unavailable. This section will review the literature on biological environmental factors and their influence on the bioavailability, uptake, and toxicity of Pb.

Van Hattum et al. (1996) studied the influence of abiotic variables, including DOC on Pb concentrations in freshwater isopods (*Proasellus meridianus* and *Asellus aquaticus*). They found that BCFs were significantly negatively correlated with DOC concentrations. Thus, as DOC concentrations increased, BCFs decreased in *P. meridianus* and *A. aquaticus*, indicating that DOC acts to inhibit the availability of Pb to these isopods.

Kruatrachue et al. (2002) investigated the combined effects of Pb and humic acid on total chlorophyll content, growth rate, multiplication rate, and Pb uptake of common duckweed. When humic acid was added to the Pb-nitrate test solutions (50, 100, and 200 mg $Pb(NO_3)_2$/ L), toxicity of Pb to *L. minor* was decreased. The addition of humic acid to the Pb-nitrate solution increased the pH. The authors suggested that there was a proton dissociation from the carboxyl group in the humic acid that complexed with Pb, resulting in a decrease in free Pb ions available to the plant.

Schwartz et al. (2004) collected natural organic matter (NOM) from several aquatic sites across Canada and investigated the effects of NOM on Pb toxicity in rainbow trout (*Oncorhynchus mykiss*). They also looked at toxicity effects as they related to the optical properties of the various NOM samples. The results showed that NOM in test water almost always increased LT_{50} and that optically dark NOM tended to decrease Pb toxicity more than did optically light NOM in rainbow trout.

In summary, non-living constituents of biological origin in the environment have been shown to reduce Pb availability and, therefore, toxicity in some aquatic organisms. It is generally thought that this occurs through complexation, or chelation processes that take place in the water column.

8.2.4.5.4 Physical Environmental Factors

This section reviews the literature on physical environmental factors and their influence on the bioavailability, uptake, and toxicity of Pb in aquatic organisms. These factors are discussed with regard to their influence individually and in combination.

Studies generally agree that as pH increases, the toxicity of Pb decreases (Horne and Dunson, 1995b; MacDonald et al., 2002). As pH decreases, Pb becomes more soluble and more readily bioavailable to aquatic organisms (Weber, 1993). Significantly lower survival, decreased hatching success, slower development, and increased egg mass and larval mortality were observed in Jefferson salamanders (*Ambystoma jeffersonianum*) and wood frogs (*Rana sylvatica*) exposed to Pb at a pH of 4.5 versus a pH of 5.5 (Horne and Dunson, 1995b). Contradictory results have been reported for invertebrates. Over a 96-h exposure period, mortality increased with decreasing pH for the bivalve *Pisidium casertanum*, while pH-independent mortality was reported for gastropods and crustacea under similar exposure conditions (Mackie, 1989). Cladocerans (*C. dubia*) and amphipods (*H. azteca*) were also more sensitive to Pb toxicity at pH 6 to 6.5 than at higher pH levels (Schubauer-Berigan et al., 1993). Lead was 100 times more toxic to the amphipod *Hyalella azteca* at a pH range of 5.0 to 6.0 (Mackie, 1989) than at a pH range of 7.0 to 8.5 (Schubauer-Berigan et al., 1993). Lead was also more toxic to fathead minnows at lower pH levels (Schubauer-Berigan et al., 1993).

The influence of pH on Pb accumulation has also been observed in sediments. Accumulation of Pb by the isopod *Asellus communis* was enhanced at low pH, after a 20-day exposure to Pb-contaminated sediments (Lewis and McIntosh, 1986). In *A. aquaticus*, temperature increases were found to be more important than increased pH in influencing Pb accumulation (Van Hattum et al., 1996). Increased water temperature was also found to reduce Pb uptake fluxes in green microalga (*Chlorella kesslerii*) (Hassler et al., 2004). Lead and zinc body concentrations in *Asellus* sp. were found to vary markedly with seasonal temperature changes, with greater concentrations present in spring and summer (Van Hattum et al., 1996).

Acute and chronic toxicity of Pb increases with decreasing water hardness, as Pb becomes more soluble and bioavailable to aquatic organisms (Horne and Dunson, 1995a; Borgmann et al., 2005). There is some evidence that water hardness and pH work

together to increase or decrease the toxicity of Pb. Jefferson salamanders exposed to Pb for 28 days at low pH and low water hardness experienced 50% mortality, while exposure to Pb at high pH and high water hardness resulted in 91.7% survival (Horne and Dunson, 1995a). Exposure to Pb at high pH and low water hardness or low pH and high water hardness resulted in 75 and 41.7% survival, respectively (Horne and Dunson, 1995a). Similar results were reported for Jefferson salamanders during a 7-day exposure and wood frogs during 7- and 28-day exposures (Horne and Dunson, 1995c). In some cases, water hardness and pH in the absence of Pb have been shown to affect survival adversely. Mean acute survival of wood frogs and Jefferson salamanders exposed to low pH and low water hardness, in the absence of Pb, was 83.3 and 91.7%, respectively. Mean chronic survival of wood frogs and Jefferson salamanders exposed to low pH and low water hardness, in the absence of Pb, was 79.2 and 41.7%, respectively (Horne and Dunson, 1995c).

High Ca^{2+} concentrations have been shown to protect against the toxic effects of Pb (Sayer et al., 1989; MacDonald et al., 2002; Hassler et al., 2004; Rogers and Wood, 2004). Calcium affects the permeability and integrity of cell membranes and intracellular contents (Sayer et al., 1989). As Ca^{2+} concentrations decrease, the passive flux of ions (e.g., Pb) and water increases. At the lowest waterborne Ca^{2+} concentration (150 µmol/L), Pb accumulation in juvenile rainbow trout (*Oncorhynchus mykiss*) branchials significantly increased as Pb concentration in water increased (Rogers and Wood, 2004). At higher Ca^{2+} concentrations, Pb accumulation did not significantly increase with Pb concentration in water. This result demonstrates the protective effects of waterborne Ca^{2+} and supports the suggestion that the Ca^{2+} component of water hardness determines the toxicity of Pb to fish (Rogers and Wood, 2004). Rogers and Wood (2004) reported that the uptake of Ca^{2+} and Pb^{2+} involves competitive inhibition of apical entry at lanthanum-sensitive Ca^{2+} channels and interference with the function of the ATP-driven baso-lateral Ca^{2+} pump. High mortality was reported in brown trout (*Salmo trutta*) fry exposed to Pb at a waterborne Ca^{2+} concentration of 20 µmol/L, while negligible mortality was reported at the same Pb concentration but at a waterborne Ca^{2+} concentration of 200 µmol/L (Sayer et al., 1989). Adverse effects to mineral uptake and skeletal development were observed in the latter test group (Sayer et al., 1989).

The bioavailability of Pb and other metals that can be simultaneously extracted in sediments may be modified through the role of acid volatile sulfide (AVS) under anoxic conditions (Tessier and Campbell, 1987; Di Toro et al., 1992; Casas and Crecelius, 1994). The term AVS (iron sulfide is an example) refers to the fraction of the sediment that consists of a reactive pool of solid-phase sulfide. This phase is available to bind divalent metals that then become unavailable for uptake by aquatic biota. The models proposed by Di Toro et al. (1992) and Casas and Crecelius (1994) predict that when the molar ratio of simultaneously extractable metals (SEM) to AVS in sediments is less than one, the metals will not be bioavailable due to complexation with available sulfide.

Salinity is an important modifying factor to metal toxicity. Verslycke et al. (2003) exposed the estuarine mysid *Neomysis integer* to individual metals, including Pb, and metal mixtures under changing salinity. At a salinity of 5%, the reported LC_{50} for Pb was 1140 µg/L (95% CL = 840, 1440 µg/L). At an increased salinity of 25‰, the toxicity of Pb was substantially reduced (LC_{50} = 4274 µg/L [95% CL = 3540, 5710 µg/L]) (Verslycke et al., 2003). The reduction in toxicity was attributed to increased complexation of Pb^{2+} with Cl^- ions.

8.2.4.5.5 Nutritional Factors

The relationship between nutrition and Pb toxicity has not been thoroughly investigated in aquatic organisms. In fact, algae species are the only aquatic organisms to have been studied fairly frequently. Although nutrients have been found to have an impact on Pb toxicity, the mechanisms involved are poorly understood. It is unclear whether the relationship between nutrients and toxicity comprises organismal nutrition (the process by which a living organism assimilates food and uses it for growth and for replacement of tissues), or whether nutrients have interacted directly with Pb, inhibiting its metabolic interaction in the organism. This section reviews the little information that has been gathered from studies documenting apparent Pb-nutrition associations in aquatic organisms.

Jampani (1988) looked at the impact of various nutrients (i.e., sodium acetate, citric acid, sodium carbonate, nitrogen, and phosphates) on reducing growth inhibition in blue-green algae (*Synechococcus aeruginosus*) exposed to 200 mg/L of Pb. Exposure to

this Pb treatment concentration caused 100% mortality in algae. Results indicated that additional nitrogen, phosphates, and some carbon sources, including sodium acetate, citric acid and sodium carbonate, all protected the algae from Pb toxicity. Algae that had been starved prior to the experiment were found to be significantly more sensitive to Pb exposure. Glucose was the only nutrient tested that did not have a significant impact on Pb toxicity in *S. aeruginosus*. In a similar study by Rao and Reddy (1985) on *Scenedesmus incrassatulus*, nitrogen, phosphate and carbon sources (including glucose), all had protective effects, and reduced Pb toxicity at 300 and 400 mg Pb/L. Both studies proposed similar hypotheses regarding nutrient-Pb mechanisms that led to reduced toxicity. One hypothesis was that the nutrients were able to reverse toxic effects. The second hypothesis was that the nutrients interacted directly with Pb, in some way sequestering the metal so as to inhibit its metabolic interaction with the organism (Rao and Reddy, 1985; Jampani, 1988).

Rai and Raizada (1989) investigated the effects of Pb on nitrate and ammonium uptake as well as carbon dioxide and nitrogen fixation in *Nostoc muscorum* over a 96-h period. Test specimens were exposed to 10, 20, and 30 mg/L of Pb. At 20 mg Pb/L, nitrate uptake was inhibited by 64% after 24 h and by 39% after 96 h. Ammonium uptake was inhibited, and similarly, inhibition decreased from 72% inhibition after 24 h to 26% inhibition after 96 h of exposure. Carbon dioxide fixation and nitrogenase activity followed similar patterns, and results indicated that Pb exposure can affect the uptake of some nutrients in *N. muscorum*.

Adam and Abdel-Basset (1990) studied the effect of Pb on metabolic processes of *Scenedesmus obliquus*. They found that nitrogenase activity was inhibited by Pb nitrate, but enhanced by Pb-acetate. As photosynthetic products and respiratory substrates, carbohydrate and lipid levels were altered by Pb. Above 30 mg/L of Pb-nitrate, both macronutrients were reduced. However, Pb-acetate was found to increase carbohydrate levels. Results suggest that Pb can have an effect on macronutrients in *S. obliquus* and that effects may vary depending on the chemical species.

Simòes Gonçalves et al. (1991) studied the impact of light, nutrients, air flux, and Pb, in various combinations, on growth inhibition in the green algae *Selenastrum capricornutum*. Results indicated that at lower Pb concentrations (<0.207 mg/L) and

increased nutrient concentrations, algae release more exudates that form inert complexes with Pb anions in the water. This suggests that *S. capricornutum* can use exudates as a protection and that this protective mechanism depends on nutrient supply. These results are consistent with those of Capelo et al. (1993), who investigated uptake of nitrogen and phosphorus in the algae *Selenastrum capricornutum* over time in the absence and presence of 0.207 mg Pb/L. They found that the presence of Pb had no significant influence on the assimilation of nitrogen and phosphorus. However, they did find that in the presence of Pb, algae released higher concentrations of organics with Pb-chelating groups.

Amiard et al. (1994) investigated the impact on soft tissue Pb concentrations of various feeding regimes on oysters (*Crassostrea gigas*) during their spat rearing. They fed test groups of *C. gigas* different amounts of *Skeletonema costatum* and additional natural phytoplankton grown in test solutions. Results showed that size and food intake both negatively correlated with metal concentrations in soft tissue. The authors hypothesized that this relationship was due in part to a diluting effect of the food.

In summary, nutrients affect Pb toxicity in those aquatic organisms that have been studied. Some nutrients seem to be capable of reducing toxicity, though the mechanisms have not been well established. Exposure to Pb has not been shown to reduce nutrient uptake ability, though it has been demonstrated that Pb exposure may lead to increased production and loss of organic material (e.g., mucus and other complex organic ligands) (Capelo et al., 1993).

8.2.4.5.6 Interactions with Other Pollutants

Most of the scientific literature reviewed in this section considered how Pb and other elements combine to affect uptake and exert toxicity. Research on the interactions of Pb with complexing ligands and other physical and biological factors was more thoroughly discussed in Sections 8.2.4.5.3 and 8.2.4.5.4. Predicting the response of organisms to mixtures of chemicals is difficult (Norwood et al., 2003). For example, at low zinc concentrations, (2:1 Pb:Zn ratio) a synergistic effect was observed in the frog, *Bufo arenarum* (Herkovits and Perez-Coll, 1991). At high concentrations of zinc, an antagonistic effect was observed as Pb toxicity was reduced. This demonstrates the

complexity of metal mixture interactions as different metal concentrations, environmental conditions (e.g., temperature, pH), and other factors can cause marked changes in the effects observed (Norwood et al., 2003). In describing Pb interactions with other elements, interaction types are classified here as antagonistic, synergistic, and additive. Each of these will be discussed below with specific reference to known Pb-metal interactions and implications on Pb uptake and toxicity.

Antagonistic Interactions

When two or more metals compete for the same binding sites or interfere with transport through cell walls or membranes, the interaction is termed less than strictly additive or antagonistic. Antagonistic interactions can reduce metal bioavailability when metals are present in combination, and may lead to reduced potential for toxicity (Hassler et al., 2004). A number of elements act in an antagonistic fashion with Pb. For example, Pb is a well-known antagonist to Ca^{2+} (Niyogi and Wood, 2004; Hassler et al., 2004), which is an essential element, required for a number of physiological processes in most organisms. Lead ions have an atomic structure similar to Ca^{2+} and can be transported either actively or passively across cell membranes in place of Ca^{2+}. An example of this interaction was reported by Behra (1993a,b) where Pb was shown to activate calmodulin reactions in rainbow trout (*O. mykiss*) and sea mussel (*Mytilus* sp.) tissues in the absence of calcium. Calmodulin (CaM) is a major intracellular calcium receptor and regulates the activities of numerous enzymes and cellular processes. Allen (1994) reported that Pb can replace calcium in body structures (e.g., bones, shells); replace zinc in ALAD, which is required for heme biosynthesis; and react with sulfhydryl groups, causing conformation protein distortion and scission of nucleic acids (Herkovits and Perez-Coll, 1991). Lead is also a known antagonist to Mg^{2+}, Na^{+}, and Cl^{-} regulation in fish (Ahern and Morris, 1998; Rogers and Wood, 2003, 2004; Niyogi and Wood, 2004). Li et al. (2004) reported on the interaction of Pb^{2+} with Cd^{2+} in the context of adsorption from solution by *Phanerochaete chrysosporium*, a filamentous fungus. The authors found that cadmium uptake decreased with increasing concentration of Pb ions with Pb outcompeting Cd^{2+} for binding sites.

Synergistic Interactions

Synergism occurs when the interaction of two or more metals causes an effect that is greater than the effect observed from the individual metals themselves (Hagopian-Schlekat et al., 2001) or, put another way, a greater than the strictly additive effect of the individual metals in a mixture (Playle, 2004). Synergism is likely the result of increased bioavailability of one or more of the metal ions due to the presence of other metals (Hassler et al., 2004). Hassler et al. (2004) reported that in the presence of copper (Cu^{2+}), there was a significantly higher rate of internalization of Pb in the green algae *Chlorella kesserii*. It was suggested that Cu^{2+} may have affected organism physiology through the disruption of cell membrane integrity. This would allow increased cation (i.e., Pb^{2+}) permeability and, therefore, substantially increase internalization of Pb. Hagopian-Schlekat et al. (2001) examined the impact of individual metals and complex metal mixtures containing Cd, Cu, Ni, Zn, and Pb to the estuarine copepod *Amphiascus tenuiremis*. The copepods were exposed to metal-spiked sediment and pore water. The mixed metal sediment toxicity tests demonstrated greater than additive toxicity to *A. tenuiremis*. It was postulated that the synergism observed was due to two or more metals affecting the same biological function. Herkovits and Perez-Coll (1991) exposed *Bufo arenarum* larvae to various Pb and zinc concentrations in solution. At low zinc concentrations, (2:1 Pb:Zn ratio), a synergistic toxic effect was observed in the frog larvae relative to the effects observed from exposure to the individual metals and at higher zinc concentrations. Enhanced Pb toxicity was attributed to the interference of Pb with cellular activities due to binding with sulfhydryl polypeptides and nucleic acid phosphates (Herkovits and Perez-Coll, 1991). Allen (1994) reported on the accumulation of numerous metals and ions into specific tissues of the tilapia *Oreochromis aureus*. Tilapia exposed to low concentrations of Pb and mercury (both at 0.05 mg/L) had significantly higher concentrations of Pb in internal organs than those fish exposed to Pb alone. Similarly, low concentrations of cadmium with low concentrations of Pb caused increased uptake of Pb in certain organs (e.g., liver, brain, and caudal muscle).

Additive Interactions

The combined effects of two or more metals may result in additivity when the observed effects are greater than that observed with individual metals but equivalent to a summation of the effects from multiple metals. Lead has been shown to complex with Cl^- in aquatic systems. For example, Verslycke et al. (2003) exposed the estuarine mysid *Neomysis integer* to six different metals, including Pb, and a combined metal mixture under changing salinity conditions. At a salinity of 5%, the reported LC_{50} for Pb was 1140 µg/L (840, 1440 µg/L). At an increased salinity of 25‰, the toxicity of Pb was substantially reduced (LC_{50} = 4274 µg/L [3540, 5710 μg/L]) (Verslycke et al., 2003). This reduction in toxicity was attributed to the increased concentration of Cl^- ion due to increased salinity, in that it complexed with divalent Pb in the test system. Exposure of *N. integer* to Pb in combination with the other five metals (Hg, Cd, Cu, Zn, Ni) resulted in roughly strictly additive toxicity (Verslycke et al., 2003).

Summary of Interactions With Other Pollutants

Norwood et al. (2003) reported that in a review and reinterpretation of published data on the interactions of metals in binary mixtures (n = 15 studies), antagonistic (6) and additive interactions (6) were the most common for Pb. The complexity of the interactions and possible modifying factors makes determining the impact of even binary metal mixtures to aquatic biota difficult (Norwood et al., 2003; Playle, 2004). The two most commonly reported Pb-element interactions are between Pb and calcium and between Pb and zinc. Both calcium and zinc are essential elements in organisms and the interaction of Pb with these ions can lead to adverse effects both by increased Pb uptake and by a decrease in Ca and Zn required for normal metabolic functions.

8.2.4.6 Factors Associated with Global Climate Change

It is highly unlikely that Pb has any influence on generation of ground-level ozone, depletion of stratospheric ozone, global warming, or other indicators of global climate change. Lead compounds have relatively short residence times in the atmosphere, making it unlikely that they will reach the stratosphere, and they do not absorb infrared radiation, making them unlikely to contribute to stratospheric ozone depletion or global

warming. Also, these compounds are unlikely to have a significant interaction with ground-level nitrogen oxides or volatile organic compounds, thus precluding generation of ground-level ozone.

Approached from another viewpoint, climate change can have a major impact on the fate/behavior of Pb in the environment and, therefore, can subsequently alter organism or ecosystem responses. For example, changes in temperature regime (Q10 rule), changes in precipitation quantity and quality (e.g., acidic deposition) may influence fate, transport, uptake, and bioavailability of Pb (Syracuse Research Corporation., 1999).

8.2.4.7 Summary

There have been a number of advancements in the understanding of Pb behavior in the environment and its impact on aquatic organisms since 1986. In particular, greater knowledge of factors that influence Pb accumulation in aquatic organisms, mechanisms of detoxification and avoidance of Pb, and greater understanding of the interactions of Pb in aquatic systems. Recently, the development of the Biotic Ligand Model (BLM) and its exploration of the activity of free metal ions at biologically reactive sites (i.e., fish gill tissue) have been a large contributor to the understanding of metal speciation and movement into and effects to aquatic biota. To date, there has been no BLM model of Pb although research has been conducted on a Pb-gill binding model for rainbow trout. Further research in support of BLM model development for Pb is recommended to further our understanding of these issues.

8.2.5 Exposure/Response of Aquatic Species

8.2.5.1 Introduction

This section outlines and highlights the critical recent advancements in the understanding of the toxicity of lead to aquatic biota. The section begins with a review of the major findings and conclusions from the 1986 Pb AQCD. The following sections summarize the research conducted since 1986 on effects of lead on primary producers, consumers, and decomposers. Issues related to indirect effects (e.g., effects on predator/prey interactions, habitat alteration,) are not to be addressed.

8.2.5.2 Summary of Conclusions From the Previous Criteria Document

The 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a) reviewed data in the context of the sublethal effects of lead exposure. The document focused on describing the types and ranges of lead exposures in ecosystems likely to adversely impact domestic animals. As such, the criteria document did not provide a comprehensive analysis of the effects of lead to most aquatic primary producers, consumers, and decomposers. For the aquatic environment, general reviews of the effects of lead to algae, aquatic vertebrates, and invertebrates were undertaken. A summary of these reviews is provided below.

Algae

The 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a) reported that some algal species (e.g., *Scenedesmus* sp.) were found to exhibit physiological changes when exposed to high lead or organolead concentrations in situ. The observed changes included increased numbers of vacuoles, deformations in cell organelles, and increased autolytic activity. Increased vacuolization was assumed to be a tolerance mechanism by which lead was immobilized within cell vacuoles.

Aquatic Vertebrates

The 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a) reported that hematological and neurological responses were the most commonly reported effects in aquatic vertebrates. These effects include red blood cell destruction and inhibition of the enzyme ALAD, required for hemoglobin synthesis. At high lead concentrations, neurological responses included neuromuscular distortion, anorexia, muscle tremor, and spinal curvature (e.g., lordosis). The lowest reported exposure concentration causing either hematological or neurological effects was 8 µg/L (U.S. Environmental Protection Agency, 1986a).

Aquatic Invertebrates

Numerous studies were cited on the effects of lead to aquatic invertebrates in the 1986 Pb AQCD (U.S. Environmental Protection Agency, 1986a). In general, lead concentrations in aquatic invertebrates were found to be correlated closely with

concentrations in water rather than food. Freshwater snails were found to accumulate lead in soft tissue, often in granular bodies of precipitated lead. Mortality and reproductive effects were reported to begin at 19 µg/L for the freshwater snail *Lymnea palutris* and 27 µg/L for *Daphnia* sp.

The review of the NAAQS for Lead (U.S. Environmental Protection Agency, 1990) made only one recommendation on the sections of the 1986 Pb AQCD reporting on effects to aquatic biota. This was the need to consider the impact of water hardness on lead bioavailability and toxicity, to be consistent with the recommendations of the AWQC for the protection of aquatic life (U.S. Environmental Protection Agency, 1985).

8.2.5.3 Recent Studies on Effects of Lead on Primary Producers

Using literature published since the 1986 Pb AQCD, this section examines the toxicity of lead (individually and in metal mixtures) to algal and aquatic plant growth, its effects on metabolic processes (e.g., nutrient uptake), and its impact on primary productivity in natural systems.

Toxicity of Lead to Algae

The toxicity of lead to algal growth has been investigated for a number of species including *Chlorella vulgaris*, *Closterium acerosum*, *Pediastrum simplex*, *Scenedesmus quadricauda*, *Scenedesmu obliquus*, *Syneschoccus aeruginosus*, and *Nostoc muscorum* (Jampani, 1988; Rai and Raizada, 1989; Adam and Abdel-Basset, 1990; Fargašová, 1993; Bilgrami and Kumar, 1997). Study durations ranged from 7 to 20 days and lead nitrate was the most commonly used form of lead. Effects to algal growth (*Chlorella vulgaris*, *Closterium acerosum*, *Pediastrum simplex*, *Scenedesmus quadricauda*), ranging from minimal to complete inhibition, have been reported at lead concentrations between 100 and 200,000 µg/L (Jampani, 1988; Bilgrami and Kumar, 1997). Most studies report the percent inhibition in test groups compared to controls rather than calculating the LOEC, NOEC, or EC_{50} values. Clinical signs of lead toxicity include the deformation and disintegration of algae cells and a shortened exponential growth phase (Jampani, 1988; Fargašová, 1993). Other effects of lead block the pathways that lead to pigment

synthesis, thus affecting photosynthesis, the cell cycle and division, and ultimately result in cell death (Jampani, 1988).

From the studies reviewed, *Closterium acerosum* is the most sensitive alga species tested (Bilgrami and Kumar, 1997). Exposure of this alga to 1000 and 10,000 µg/L of lead nitrate for 6 days resulted in cell growth that was 52.6 and 17.4%, respectively, of controls (Bilgrami and Kumar, 1997). *Chlorella vulgaris*, *Pediastrum simplex*, and *Scenedesmus quadricauda* were also exposed to lead nitrate in this study. Compared to controls, cell growth at 1000 and 10,000 µg lead nitrate/L was 65.3 and 48.7%, 64.5 and 42.7%, and 77.6 and 63.2%, respectively (Bilgrami and Kumar, 1997). *Scenedesmus quadricauda* exhibited a similar magnitude of effects when exposed to lead (Pb^{2+}) for 20 days at 0, 5500, 11,000, 16,500, 22,000, 27,500, and 33,000 µg/L (Fargašová, 1993). This study reported an EC_{50} for growth inhibition at 13,180 µg/L (95% CI: 10,190, 14,620). Decreased cell number, but increased cell size, was observed in *Selenastrum capricornutum*[11] exposed to lead (Pb^{2+}) at 207.2 µg/L and a Q/V (flux of air [Q] divided by volume of the culture [V]) of 4.7×10^{-3} sec^{-1} for 9 days (Simòes Gonçalves et al., 1991). The Q/V is a measure of culture growth where an increase in the Q/V ratio indicates growth. The pigment concentration per cell decreased with exposure to lead, so while the algae cells were larger, they were less healthy (Simòes Gonçalves et al., 1991). Growth rates were not reported, making comparison with other studies difficult.

High lead concentrations were required to elicit effects in *Nostoc muscorum* and *Scenedesmus aeruginosus* (Jampani, 1988; Rai and Raizada, 1989). Following 15 days of exposure, test groups exposed to 10,000, 20,000, and 30,000 µg/L lead experienced growth rates that were 90.5, 76.9, and 66.7% of the controls (Rai and Raizada, 1989). *Synechococcus aeruginosus* experienced little inhibition of growth from exposure to lead nitrate up to a concentration of 82,000 µg/L (Jampani, 1988). At a test concentration of 100,000 µg/L, complete inhibition of growth was observed, and at a concentration of 200,000 µg/L, algae failed to establish a single colony (Jampani, 1988). *Scenedesmus obliquus* are quite tolerant to the effects of lead nitrate and lead acetate on growth.

[11]The species name *Selenastrum capricornutum* has been changed to *Pseudokirchneriella subcapitata*. The former species name is used in this report.

Algae exposed to lead nitrate or lead acetate up to 180,000 µg/L had higher cell numbers than controls (Adam and Abdel-Basset, 1990). Exposure to the highest concentration of 300,000 µg/L lead nitrate or lead acetate resulted in cell numbers that were 81 and 90% of the controls, respectively (Adam and Abdel-Basset, 1990).

Lead in combination with other metals (e.g., Pb and Cd, Pb and Ni, etc.) is generally less toxic than exposure to lead alone (Rai and Raizada, 1989). *Nostoc muscorum* exposed to chromium and lead in combination demonstrated better growth than when exposed to either of the metals alone (Rai and Raizada, 1989). Antagonistic interaction was observed in the exposure of *Nostoc muscorum* to lead and nickel in combination (Rai and Raizada, 1989). When applied separately, these metals demonstrated different levels of toxicity; however, in combination, they exerted similar effects (Rai and Raizada, 1989). More information on toxic interactions of lead with other metals is provided in Section 8.2.4.5.6.

Aquatic Plants

The toxicity of lead to aquatic plant growth has been studied using *Spirodela polyrhiza*, *Azolla pinnata*, and *Lemna gibba* (Gaur et al., 1994; Gupta and Chandra, 1994; Miranda and Ilangovan, 1996). Test durations ranged from 4 to 25 days and test concentrations ranged between 49.7 and 500,000 µg/L (Gaur et al., 1994; Miranda and Ilangovan, 1996). Research on aquatic plants has focused on the effects of lead on aquatic plant growth and chlorophyll and protein content.

Of the species reviewed here, the effects of lead on aquatic plant growth are most pronounced in *Azolla pinnata* (Gaur et al., 1994). An EC_{50} of 1100 µg/L was reported for *Azolla pinnata* exposed to lead nitrate for 4 days. *Spirodela polyrhiza* exposed to lead nitrate under the same test conditions had a reported EC_{50} for growth of 3730 µg/L (Gaur et al., 1994). *Lemna gibba* was shown to be the least sensitive plant species to lead: significant growth inhibition was reported at concentrations of 200,000 µg/L or greater after 25 days of exposure to concentrations of 30,000, 50,000, 100,000, 200,000, 300,000, or 500,000 µg/L (Miranda and Ilangovan, 1996). The maximum growth rate for *Lemna gibba* was observed at 10 days of exposure. After this point, the growth rate declined in controls and test concentrations (Miranda and Ilangovan, 1996). Clinical

signs of lead toxicity include yellowing and disintegration of fronds, reduced frond size, and chlorosis (Gaur et al., 1994; Miranda and Ilangovan, 1996). Toxicity results suggest that effects to growth from exposure to lead occur in a dose-dependent manner (Gaur et al., 1994).

Effects of Lead on Metabolic Processes

Algal and aquatic plant metabolic processes are variously affected by exposure to lead, both singularly and in combination with other metals. Lead adversely affects the metabolic processes of nitrate uptake, nitrogen fixation, ammonium uptake, and carbon fixation at concentrations of 20,000 µg/L or greater (Rai and Raizada, 1989). Lead in combination with nickel has an antagonistic effect on nitrogen fixation and ammonium uptake, but a synergistic effect on nitrate uptake and carbon fixation (Rai and Raizada, 1989). Lead in combination with chromium has an antagonistic effect on nitrate uptake, but it has a synergistic effect on nitrogen fixation, ammonium uptake, and carbon fixation (Rai and Raizada, 1989).

Lead effects on nitrate uptake in *Nostoc muscorum* (µg NO_3/µg Chl *a*) were greatest after 24 h, when exposure to 20,000 µg/L reduced nitrate uptake by 64.3% compared to controls. Nitrate uptake reported after 48, 72, and 96 h was reduced by 30.0, 37.5, and 38.9%, respectively, compared to controls (Rai and Raizada, 1989). Lead in combination with chromium, both at a test concentration of 20,000 µg/L, demonstrated antagonistic effects on nitrate uptake. Compared to controls, nitrate uptake was reduced by 52.4, 30, 25, and 22.2% at 24, 48, 72 and 96 h, respectively (Rai and Raizada, 1989). The greatest effect on uptake occurred at 24 h when, compared to controls, a 52.4% reduction was reported in the test concentration. Lead and nickel in combination at test concentrations of 20,000 and 1000 µg/L, respectively, resulted in a greater reduction of nitrate uptake than lead alone at 48, 72, and 96 h (Rai and Raizada, 1989).

After 24, 48, and 72 h of exposure to lead at 20,000 µg/L, nitrogenase activity (nmol C_2H_4/ µg protein/hr) in *Nostoc muscorum* was reduced by 39.3, 61.8, and 14.1%, respectively, compared to controls (Rai and Raizada, 1989). A concentration of 207.2 µg/L had little effect on nitrogen or phosphorus assimilation in *Selenastrum capricornutum* over 7 days (Capelo et al., 1993). An antagonistic effect on nitrogenase

activity was generally reported for *Nostoc muscorum* exposed to lead in combination with nickel at 20,000 and 1,000 µg/L, respectively (Rai and Raizada, 1989). Compared to controls, nitrogenase activity was reduced by 42.9, 32.7, and 13.6% at 24, 48, and 72 h, respectively (Rai and Raizada, 1989). Lead and chromium, both administered at a concentration of 20,000 µg/L, had a synergistic impact on nitrogenase activity in *Nostoc muscorum*. Nitrogenase activity in the test group was reduced by 60.7, 60, and 50% compared to the controls at 24, 48, and 72 h, respectively (Rai and Raizada, 1989).

Lead-induced inhibition of ammonium uptake (µg NH_4 uptake/µg Chl *a*) was greatest in *Nostoc muscorum* after 48 h of exposure to 20,000 µg/L of lead. Compared to controls, the lead test concentration 20,000 µg/L reduced ammonium uptake by 72, 82, 61, and 26 % at 24, 48, 72, and 96 h, respectively (Rai and Raizada, 1989). Lead in combination with nickel at concentrations of 20,000 and 1,000 µg/L, respectively, demonstrated an antagonistic effect on ammonium uptake. Compared to controls, ammonium uptake in the test group was reduced by 44.9, 54.1, 23.3, and 4% at 24, 48, 72, and 96 h, respectively (Rai and Raizada, 1989). Lead in combination with chromium, both at concentrations of 20,000 µg/L, demonstrated a synergistic interaction with 24, 48, 72, and 96 h uptake rates reduced by 87.2, 88.5, 72.5, and 50 %, respectively, compared to controls (Rai and Raizada, 1989).

Nostoc muscorum exposed to 20,000 µg/L of lead experienced the greatest reduction in carbon fixation at 0.5 h of exposure: 62% compared to controls. Inhibition of carbon fixation in the test group was less pronounced after 1 and 2 h of exposure: 29 and 13% of controls (Rai and Raizada, 1989). Lead in combination with nickel or chromium had synergistic effects to carbon fixation. Lead and nickel concentrations of 20,000 and 1000 µg/L, respectively, resulted in 0.5, 1, and 2 h carbon fixation rates reduced by 93, 92, and 91%, respectively, compared to controls (Rai and Raizada, 1989). Lead with chromium at concentrations of 20,000 µg/L resulted in 0.5, 1, and 2 h carbon fixation rates reduced by 65, 58, and 50%, respectively, compared to controls.

Nutrients such as nitrogen, phosphate, sodium acetate, sodium carbonate, and citric acid have been shown to protect against the toxic effects of lead to algae (Jampani, 1988). Nitrogen compounds (ammonium chloride, potassium nitrate, sodium nitrate, sodium nitrite) protected *Synechococcus aeruginosus* from a lethal lead nitrate dose of

200,000 µg/L (Jampani, 1988). Two phosphates (K_2HPO_4 and Na_2HPO_4) were found to improve *Synechococcus aeruginosus* survival from 0 to 72% at 200,000 µg/L of lead nitrate (Jampani, 1988).

Compared to controls, protein content was reduced by 54.2 and 51.9% in aquatic plants *Vallisneria spiralis* and *Hydrilla verticillata*, respectively, exposed to lead for 7 days at 20,720 µg/L (Gupta and Chandra, 1994). Decreased soluble protein content has been observed in *Scenedesmus obliquus* exposed to lead nitrate or lead acetate at concentrations greater than 30,000 µg/L, and in *Lemna gibba* at concentrations greater than 200,000 µg/L (Adam and Abdel-Basset, 1990; Miranda and Ilangovan, 1996). *Lemna gibba* also showed increased loss of soluble starch at concentrations >200,000 µg/L (Miranda and Ilangovan, 1996). Under the conditions described previously (Gupta and Chandra, 1994), EC_{50} values for chlorophyll content were 14,504 and 18,648 µg/L for *Vallisneria spiralis* and *Hydrilla verticillata*, respectively (Gupta and Chandra, 1994). Effects to chlorophyll *a* content have been observed in *Scenedesmus obliquus* at lead nitrate and lead acetate concentrations >30,000 µg/L (Adam and Abdel-Basset, 1990).

Summary of Toxic Effects Observed in Single-Species Bioassays

Algae and aquatic plants have a wide range in sensitivity to the effects of lead in water. Both groups of primary producers experience EC_{50} values for growth inhibition between approximately 1000 and >100,000 µg/L (Jampani, 1988; Gaur et al., 1994; Bilgrami and Kumar, 1997). The most sensitive primary producers reported in the literature for effects to growth were *Closterium acersoum* and *Azolla pinnata* (Gaur et al., 1994; Bilgrami and Kumar, 1997). The least sensitive primary producers reported in the literature for effects to growth were *Synechococcus aeruginosus* and *Lemna gibba* (Jampani, 1988; Miranda and Ilangovan, 1996). Exposure to lead in combination with other metals is generally less toxic to growth than exposure to lead alone. Studies have shown that lead adversely affects the metabolic processes of nitrate uptake, nitrogen fixation, ammonium uptake, and carbon fixation (Rai and Raizada, 1989). Lead in combination with nickel or chromium produced synergistic effects for nitrate uptake, nitrogenase activities, ammonium uptake, and carbon fixation (Rai and Raizada, 1989).

Leads Effects on Primary Productivity

Lead nitrate and lead acetate have been shown to have adverse effects on the primary productivity of aquatic plants in two water bodies in India (Jayaraj et al., 1992). One of the two water bodies was a freshwater tank that receives wastewater and supports a rich population of hyacinths, and the other was a wastewater stabilization pond. Water quality characteristics in the freshwater tank were pH = 7.5, dissolved oxygen = 6 mg/L, and water hardness ($CaCO_3$) = 100 mg/L. Water quality characteristics in the wastewater pond were pH = 8.1, dissolved oxygen = 6.2 mg/L, and water hardness ($CaCO_3$) = 160 mg/L (Jayaraj et al., 1992). Lead nitrate concentrations of 500, 5000, 10,000, 25,000, and 50,000 µg/L were combined with appropriate water samples in light and dark bottles and suspended in each of the water bodies for 4 h. The concentrations of lead acetate (5000, 10,000, 25,000, 50,000, and 100,000 µg/L) were applied in the same manner. The EC_{50} values were determined based on the concentration required to inhibit gross productivity (GP) and net productivity (NP) by 50% (Jayaraj et al., 1992). The results demonstrated that lead nitrate was more toxic to primary production than lead acetate. In the freshwater tank, lead nitrate EC_{50} values for GP and NP were 25,100 and 6310 µg/L, respectively, compared to lead acetate EC_{50} values of 50,100 and 28,200 µg/L for GP and NP, respectively (Jayaraj et al., 1992). In the stabilization pond, lead nitrate EC_{50} values for GP and NP were 31,600 and 28,200 µg/L, respectively, compared to lead acetate EC_{50} values of 79,400 and 316 µg/L for GP and NP, respectively (Jayaraj et al., 1992). The higher toxicity reported in the freshwater tank was attributed to differences in species composition and diversity. The freshwater tank was dominated by water hyacinths that decreased the photic zone available for photosynthesis and consumed a great deal of available nutrients. The stabilization pond had a rich nutrient budget, resulting in improved alga growth and species diversity (Jayaraj et al., 1992).

8.2.5.4 Recent Studies on Effects of Lead on Consumers

This section focuses on the effects of lead to aquatic biota including invertebrates, fish, and other biota with an aquatic life stage (e.g., amphibians). It is not intended to be a comprehensive review of all research conducted. Rather, the intent is to illustrate the effects Pb can have on freshwater and marine aquatic species. Eisler (2000) provides an

overview of much of the recent available literature on the toxicity of lead to fish and aquatic invertebrates. An extensive literature search was conducted using numerous electronic bibliographic and database services (e.g., DIALOG, EPA ECOTOX) and limited temporally from 1986 to present. This temporal limit was due to the availability of the EPA water quality criteria report for the protection of aquatic life, released in 1986 (U.S. Environmental Protection Agency, 1986b). Based on the results of the literature search and recent reviews of the toxicity of lead (Eisler, 2000), numerous studies have been published on the toxicity of lead to aquatic consumers. Hardness, pH, temperature, and other factors are important considerations when characterizing the acute and chronic toxicity of lead (Besser et al., 2005) (Section 8.2.4.5). However, many of the studies reviewed did not report critical information on control mortality, water quality parameters, or statistical methods, making comparing effects between studies difficult. Studies reporting only physiological responses to lead exposure (e.g., reduction of ALAD) are not discussed here, as this topic was covered more completely in Section 8.2.4.4. This section provides a review of toxicity studies conducted with invertebrates, fish, and other aquatic organisms.

Invertebrates

Exposure of invertebrates to Pb can lead to adverse effects on reproduction, growth, survival, and metabolism (Eisler, 2000). The following presents information on the toxicity of lead to invertebrates in fresh and marine waters.

Freshwater Invertebrates

Acute and chronic lead toxicity data for freshwater invertebrates are summarized in Table 8-2.5.1. As described in Section 8.2.4.5.4, water hardness is a critical factor governing the solubility, bioavailability, and ultimately the toxicity of lead. The acute and chronic toxicity of lead increases with decreasing water hardness as lead becomes more soluble and bioavailable to aquatic organisms. For example, Borgmann et al. (2005) examined the toxicity of 63 metals, including Pb, to *Hyalella azteca* at two levels of water hardness (soft water hardness, 18 mg $CaCO_3$/L; hard water, 124 mg $CaCO_3$/L). Lead was 23 times more acutely toxic to *H. azteca* in soft water than hard water. Besser et al. (2005) found that acute toxicity to *H. azteca* was also modified by water hardness.

Table 8-2.5.1. Effects of Lead to Freshwater and Marine Invertebrates

Species	Chemical	Endpoint: Conc. (µg/L)*	Duration of Exposure	Water Chemistry	Test Type - Effect	Reference
Freshwater						
Cladoceran *(Ceriodaphnia dubia)*	lead chloride	LC_{50}: 280 >2,700 >2,700	48 h	pH: 6–6.5 7–7.5 8–8.5 Hardness: 280-300 mg/L $CaCO_3$	static-survival	Schubauer-Berigan et al. (1993)
Worm (*Lumbriculus variegatus*)	lead chloride	LC_{50}: >8,000 >8,000 >8,000	96 h	pH: 6–6.5 7–7.5 8–8.5 Hardness: 280-300 mg/L $CaCO_3$	static-survival	Schubauer-Berigan et al. (1993)
Amphipod (*Hyalella azteca)*	lead chloride	LC_{50}: <90 >5,400 >5,400	96 h	pH: 6–6.5 7–7.5 8–8.5 Hardness: 280-300 mg/L $CaCO_3$	static-survival	Schubauer-Berigan et al. (1993)
Amphipod (*Hyalella azteca*)	lead chloride	LC_{50}: 27 (20.1-36.4)	8 days	Hardness 130 mg/L pH 7.8-8.6	renewal, 1-week-old amphipods	MacLean et al. (1996)
Amphipod (*Hyalella azteca*)	lead chloride	LC_{50}: 60 (53.6-67.3)	8 days	Hardness 130 mg/L pH 7.8–8.6	renewal, 10- to 16-week old amphipods	MacLean et al. (1996)
Mayfly (*Leptophlebia marginata)*	lead chloride	LC_{50}: 1090 (400-133200)	96 h	pH: 4.5	acute - survival	Gerhardt (1994)
Mayfly (*Leptophlebia marginata*)	lead chloride	LC_{50}: 5000	96 h	pH 7.0	acute - survival	Gerhardt (1994)

Table 8-2.5.1 (cont'd). Effects of Lead to Freshwater and Marine Invertebrates

Species	Chemical	Endpoint: Conc. (µg/L)*	Duration of Exposure	Water Chemistry	Test Type - Effect	Reference
Amphipod (*Hyalella azteca*)	lead nitrate	LC_{50}: 10 21 18	96 h	pH: 5.0 5.5 6.0	acute-survival	Mackie (1989)
Bivalve (*Pisidium compressum*)	lead nitrate	LC_{50}: 38,000 21,300 11,400	96 h	pH: 3.5 4.0 4.5	acute- survival	Mackie (1989)
Bivalve (*Pisidium casertanum*)	lead nitrate	LC_{50}: 23,600 23,500 56,000	96 h	pH: 3.5 4.0 4.5	acute- survival	Mackie (1989)
Gastropod (*Amnicola limosa)*	lead nitrate	LC_{50}: 10,300 20,600 9,500	96 h	pH: 3.5 4.0 4.5	acute- survival	Mackie (1989)
Mussel (*Dreissena polymorpha*)	lead nitrate	EC_{50}: 370 91	48 h 10 wks	pH = 7.9; Hardness = 150 mg $CaCO_3$/L; Temp = 15 °C	renewal - filtration	Kraak et al. (1994)
Mussel (*Dreissena polymorpha*)	lead nitrate	LT_{50}: 358	72 days	pH = 7.9; Hardness = 150 mg $CaCO_3$/L; Temp = 15 °C	renewal - filtration	Kraak et al. (1994)
Amphipod (*Hyalella azteca*)	lead nitrate	LC_{50}: 4.8 (3.3 - 7.1)	7 days	pH = 7.37 - 8.27 Hardness = 18 mg $CaCO_3$/L DOC = 0.28 mg/L	renewal - survival	Borgmann et al. (2005)
Amphipod (*Hyalella azteca*)	lead nitrate	LC_{50}: 113 (101 -126)	7 days	pH = 8.21 - 8.46 Hardness = 124 mg $CaCO_3$/L; DOC = 1.1 mg/L	renewal - survival	Borgmann et al. (2005)

Table 8-2.5.1 (cont'd). Effects of Lead to Freshwater and Marine Invertebrates

Species	Chemical	Endpoint: Conc. (μg/L)*	Duration of Exposure	Water Chemistry	Test Type - Effect	Reference
Mayfly (*Leptophlebia marginata*)	lead chloride	LC_{50}: 1090 (95% C.I. = 133.2) >5000	96 h	pH = 4.5 - 6.5; DOC - 21.6 mg Cl^{-1}; Cond = 7.0 μS cm^{-1}	renewal - survival	Gerhardt (1994)
Cladoceran (*D. magna*)	lead nitrate	LC_{50}: 0.45	48 h	pH = 8.3± 0.2 Hardness ($CaCO_3$) = 150 mg/L Temp= 20 °C	static - embryogenesis	Bodar et al. (1989)
Cladoceran (*D. magna*)	lead chloride	NOEC: 260	12 to 21 d	Not specified	renewal - reproduction	Enserink et al. (1991)
Cladoceran *D.magna*)	lead chloride	NOEC: 270	10 d	Not specified	renewal - growth	Enserink et al. (1991)
Amphipod (*Hyalella azteca*)	lead	LOEC: (Dissolved Pb) 192 (Total Pb) 466	96 h	pH = 8.27 Hardness ($CaCO_3$) = 275 mg/L Temp = 21.1 °C	flow through - survival	Besser et al. (2005)
Tubificid worm (*Tubifex tubifex*)	lead nitrate	EC_{50}: 237 (183–316) 142 (107–184)	24 h 48 h	pH = 7.5–7.7 Hardness = 245 mg/L Temp = 29.5–31 °C	static - immobilization	Khangarot (1991)
Marine						
Copepod (*Amphiascus tenuiremis*)	lead	LC_{50}: sediment 2462 µg metal/dry sediment	96 h	pH = 7.7 ± 0.1 Dissolved O_2 -6.3 ± 0.3 mg/L Salinity – 32 ppt		Hagopian-Schlekat et al. (2001)
Bivalve (*Mytilus galloprovincialis*)	lead nitrate	EC_{50} : 221 (58.9–346.3) LOEC : 50		artificial seawater	embryogenesis	Beiras and Albentosa (2003)

* - Brackets after effect concentration are 95% confidence intervals.

At a mean pH of 7.97 in soft water (hardness ($CaCO_3$) = 71 mg/L) mortality was >50% for *H. azteca* at a dissolved Pb concentration of 151 µg/L. The LOEC for survival in hard water (hardness ($CaCO_3$) = 275 mg/L) at pH 8.27 was 192 µg/L as dissolved Pb and 466 µg/L as total Pb. Both waterborne and dietary Pb were found to contribute to reduced survival of *H. azteca* (Besser et al., 2005).

Exposure duration may also play an important role in lead toxicity in some species. For example, Kraak et al. (1994) reported that filtration in the freshwater mussel *Dreissena polymorpha* was adversely affected at significantly lower Pb concentrations over 10 weeks of exposure than was the case after 48 h of exposure.

The influence of pH on lead toxicity in freshwater invertebrates varies between invertebrate species. Over a 96-h exposure period, mortality increased with decreasing pH in the bivalve *Pisidium casertanum*, while pH-independent mortality was reported for gastropod and crustacean species under similar exposure conditions (Mackie, 1989). Cladocerans (*Ceriodaphnia dubia*), amphipods (*H. azteca*), and mayflies (*Leptophlebia marginata*) were also more sensitive to lead toxicity at lower pH levels (Schubauer-Berigan et al., 1993; Gerhardt, 1994). Lead was 100 times more toxic to the amphipod*, Hyalella azteca*, at a pH range of 5.0 to 6.0 (Mackie, 1989) than at a pH range of 7.0 to 8.5 (Schubauer-Berigan et al., 1993).

The physiology of an aquatic organism at certain life stages may be important when determining the toxicity of metals to test organisms. For example, Bodar et al. (1989) exposed early life stages of *Daphnia magna* to concentrations of $Pb(NO_3)_2$. The test medium had a pH of 8.3 ± 0.2, water hardness ($CaCO_3$) of 150 mg/L, and temperature of 20 ± 1 °C. Lead concentrations of <100 mg/L had no impact on *Daphnia* egg development. The authors suggested that this may due to the *Daphnia* egg structure, which consists of two layers: the inner vitelline layer and outer chlorion layer. The chlorion layer in other species (e.g., rainbow trout) is known to adsorb metals, thereby, preventing ionic injury to the developing embryo.

Exposures to sediment-associated lead can be toxic to sediment-dwelling organisms. In freshwater sediments, 48-h exposure of water fleas (*Daphnia magna*) to 7000 mg/kg dw significantly reduced mobility, while exposure to 13,400 mg/kg dw for 24 h produced the same effect (Dave, 1992a,b). Longer-term (i.e., 14-day) exposure of midges (*Chironomus tentans*) to sediments containing 31,900 mg/kg dw of lead resulted in 100% mortality.

Marine Invertebrates

In estuarine environments, salinity is an important modifying factor to Pb toxicity. Verslycke et al. (2003) exposed the estuarine mysid *Neomysis integer* to individual metals, including Pb, and metal mixtures under changing salinity. Water temperature (20 ± 1 ^{o}C) and salinity were reported, although no other water quality parameters were available (e.g., pH, water hardness). At a salinity of 5‰, the reported LC_{50} for Pb was 1140 µg/L (95% CI: 840, 1440 µg/L). At an increased salinity of 25‰, the toxicity of lead was substantially reduced (LC_{50} = 4274 µg/L [3540, 5710 µg/L]) (Verslycke et al., 2003).

Sensitivity to Pb can also vary between genders in some aquatic organisms. For example, Hagopian-Schlekat et al. (2001) examined the toxicity of lead chloride in sediment and sediment pore water to female and male estuarine copepods *Amphiascus tenuiremis*. The reported LC_{50} for total lead was 2462 mg Pb/kg dw (95% CI: 2097, 2891 mg Pb/kg dw). Gender effects were observed in that male copepods were more sensitive ($p = 0.038$) to Pb than females as determined by generalized linear model analysis.

Beiras and Albentosa (2003) examined the inhibition of embryo development in commercial bivalves *Ruditapes decussatus* and *Mytilus galloprovincialis* after exposure to concentrations of $Pb(NO_3)_2$ in seawater. No water chemistry parameters other than temperature were reported (test conducted at 20 ^{o}C). An EC_{50} range for *R. decussatus* was reported as 156 to 312 µg/L, as insufficient data were available to calculate the actual EC_{50}. The lowest observable effect concentration (LOEC) was 156 µg/L. For *M. galloprovincialis*, the EC_{50} was 221 µg/L (95% CI: 58.9, 346.3) while the LOEC was reported as 50 µg/L.

Fish

The general symptoms of lead toxicity in fish include production of excess mucus, lordosis, anemia, darkening of the dorsal tail region, degeneration of the caudal fin, destruction of spinal neurons, ALAD inhibition, growth inhibition, renal pathology, reproductive effects, growth inhibition, and mortality (Eisler, 2000). Toxicity in fish has been closely correlated with duration of exposure and uptake (Eisler, 2000). The following presents information on the toxicity of lead to fish in fresh and marine waters. Table 8-2.5.2 summarizes the effects of Pb on freshwater and marine fish.

Table 8-2.5.2. Effects of Pb to Freshwater and Marine Fish

Species	Chemical	Endpoint: Conc. (μg/L)	Duration of Exposure	Water Chemistry	Comments	Reference
Freshwater						
Fathead minnow (*Pimephales promelas*)	lead chloride	LC_{50}: 810 >5,400 >5,400	96 h	pH: 6–6.5 7–7.5 8–8.5 Hardness: 280-300 mg/L $CaCO_3$	static, measured	Schubauer-Berigan et al. (1993)
Rainbow trout - mature males (*Oncorhynchus mykiss*)	lead nitrate	Reproductive effects: 10	12 days	Hardness 128 mg/L $CaCO_3$	Decreased spermatocyte development	Ruby et al. (1993)
Fathead minnow (*Pimephales promelas*)	lead acetate	Reproductive Effects: 500	29 days	pH: 7.5–8.5; Hardness 130 mg/L $CaCO_3$; 22–25 °C (Pb 95% soluble)	Fewer viable eggs produced, testicular damage	Weber (1993)
Rainbow trout – Juvenile (*Oncorhynchus mykiss*)	lead nitrate	LC_{50}: 1000 (800 - 1400)	96 h	pH: 7.9–8.0 DOC = 3 mg/L Hardness ($CaCO_3$) = 140 mg/L	Flow through - Survival	Rogers and Wood (2003)
Common carp (*Cyprinus carpio*)	not reported	LC_{50}: 6.5 cm fish – 1030 3.5 cm fish – 300	96 h	pH: 7.1 Temperature–15 °C Oxygen sat. 6.4 mg/L	static-renewal - Survival	Alam and Maughan (1995)

Freshwater Fish

Many of the toxicity modifying factors described above (e.g., pH, DOC) for invertebrates are also important modifying factors for lead toxicity to fish species. The effects of pH on lead bioavailability and subsequent toxicity have been well studied (Sayer et al., 1989; Spry and Wiener, 1991; Schubauer-Berigan et al., 1993; Stouthart et al., 1994; MacDonald et al., 2002; Rogers and Wood, 2003). Schubauer-Berigan et al. (1993) exposed fathead minnow to lead chloride over 96 hours. The reported LC_{50} ranged from 810 to >5400 µg/L at pH 6 to 6.5 and pH 7 to 8.5, respectively.

Water hardness also has a strong influence on the effects of lead to fish. Chronic exposure of rainbow trout fry to lead in soft water resulted in spinal deformities at 71 to 146 µg/L after 2 months of exposure (Sauter et al., 1976) or 13.2 to 27 µg/L (Davies and Everhart, 1973; Davies et al., 1976), after 19 months of exposure. When exposed to lead in hard water, only 0 and 10% of the trout (*Oncorhynchus mykiss*) developed spinal deformities at measured lead concentrations of 190 and 380 µg/L, respectively. In soft water, 44 and 97% of the trout developed spinal deformities at concentrations of 31 and 62 µg/L, respectively (Davies et al., 1976). The maximum acceptable toxicant concentration (MATC) for rainbow trout fry in soft water was 4.1 to 7.6 µg/L (Davies et al., 1976), while the MATC for brook trout was 58 to 119 µg/L (Holcombe et al., 1976). Histological reproductive abnormalities were noted in mature male rainbow trout at 10 µg/L lead nitrate (Ruby et al., 1993).

Schwartz et al. (2004) examined the influence of NOM on lead toxicity to rainbow trout exposed for 96 h in a static system. The pH of the exposure system ranged between 6.5 and 7.0, temperature was maintained between 9 and 11 °C, and lead was added as $PbCl_2$. NOM from a number of U.S. rivers and lakes was then added to the test system, and the LT_{50} was reported. NOM was found to reduce the toxic effects of Pb to rainbow trout.

Fish size is an important variable in determining the adverse effects of lead. Alam and Maughan (1995) exposed two different sizes of common carp (*Cyprinus carpio*) to lead concentrations to observed effects on carp mortality. Water chemistry parameters were reported (pH = 7.1; temperature = 20 °C). Smaller fish (3.5 cm) were found to be more sensitive to Pb than were larger fish (6.5 cm). The reported LC_{50}s were 0.44 mg/L and 1.03 mg/L, respectively.

Marine Fish

There were no studies available that examined the toxicity of lead to marine fish species for the time period examined (1986 to present). However, Eisler (2000) reviewed available research on lead toxicity to marine species and reported studies done prior to 1986. Acute toxicity values ranged from 50 µg/L to 300,000 µg/L in plaice (*Pleuronectes platessa*) exposed to organic and inorganic forms of lead (Eisler, 2000). Organolead compounds (e.g., tetramethyl lead, tetraethyl lead, triethyl lead, diethyl lead) were generally more toxic to plaice than inorganic lead (Maddock and Taylor, 1980).

Other Aquatic Biota

A paucity of data exist on the effects of lead to growth, reproduction, and survival of aquatic stages of frogs and turtles. Rice et al. (1999) exposed frog larvae (*Rana catesbeiana*) to 780 µg Pb/L and two oxygen concentrations (3.5 or 7.85 mg/L) for 7 days (Table 8-2.5.3). Exposure conditions included water hardness of 233 to 244 mg $CaCO_3$/L, pH from 7.85 to 7.9, and temperature at 23 °C. Frog larvae were found to display little to no activity in the low oxygen and high Pb treatment. Hypoxia-like behavior was exhibited in larvae exposed to both low and high oxygen concentrations and high Pb. Therefore, larvae of *R. catesbeiana* showed sensitivity to Pb and responded with hypoxia-like behavior. Additionally, the larvae in the lead treatment were found to have lost body mass relative to controls and the other treatments. Rice et al. (1999) suggested that the decrease in mass likely indicated the beginning of a period of reduced growth rate. Larvae exposed for longer periods (>4 weeks) were smaller and metamorphosed later compared to unexposed individuals.

Herkovits and Pérez-Coll (1991) examined lead toxicity to amphibian larvae (*Bufo arenarum*). Larvae (n = 50) were exposed for up to 120 h at two Pb concentrations, 8 mg Pb^{2+}/L and 16 mg Pb^{2+}/L. Relative to controls, the 8 mg Pb^{2+}/L treatment group exhibited 40% mortality and the 16 mg Pb^{2+}/L group 60% mortality after 120 h ($p < 0.05$). The authors reported behavioral effects, erratic swimming, and loss of equilibrium during the tests, symptoms that are consistent with the action of lead on the central and peripheral nervous systems (Rice et al., 1999).

Table 8-2.5.3. Nonlethal Effects in Amphibians

Species	Chemical	Endpoint: Concentration	Duration of Exposure	Water Chemistry	Comments	Reference
Frogs (*Rana ridibunda*)	lead nitrate	Biochemical effects: 14,000 µg/L	30 days	not specified	Hepatic ALAD decreased by 90%	Vogiatzis and Loumbourdis (1999)
Frogs (*Bufo arenarum*)		Mortality: 16 mg Pb2+/L	5 days	not specified	Effects reported include erratic swimming, loss of equilibrium	Herkovits and Perez-Coll (1991)
Frogs (*Rana catesbeiana*)		Hypoxia-like behavior: 780 µg/L	7 days	O_2 = 3.5-7.85 mg/L pH = 7.85-7.9 Temp = 23 °C $CaCO_3$ = 233-244 mg/L	Larvae used	Rice et al. (1999)
Turtle Hatchlings (*Trachemys scripta*)	lead acetate	NOEL: 100 µg/g (Survival and behavior)	4 weeks	N/A	Exposure via single injection	Burger et al. (1998)

Behavior (i.e., righting, body turnover, seeking cover), growth, and survival of hatchling slider turtles (*Trachemys scripta*) exposed to lead acetate were investigated in one study (Burger et al., 1998). In the first part of the study, 6-month-old hatchlings received single lead acetate injections at 50 or 100 µg/g body weight (bw). In the second part of the study, 3-week-old turtles were injected once with doses of 250, 1000 or 2500 µg/g bw. There were no differences in survival, growth, or behavior for hatchlings in the first study, however, several effects were reported from the second part of the study at doses in the range of 250 to 2,500 µg/g bw. As the dose increased, so did the plastron length (i.e., ventral section of the shell), carapace length, and weight. The highest dose group had the lowest survival rate with an LD_{50} of 500 µg/g bw. Behavioral effects included slower times of righting behavior and seeking cover. The authors suggested a NOEL of 100 µg/g bw for slider turtles for survival and behavior.

8.2.5.5 Recent Studies on Effects of Lead on Decomposers

In this section, decomposers are defined as being bacteria and other microorganisms. Many invertebrates are also potentially considered decomposers, but the effects of lead to invertebrates have been described in previous sections. There were no toxicity studies located on the effects of lead to aquatic decomposers in the time period of interest.

8.2.5.6 Summary

Lead in all its forms is known to cause adverse effects in aquatic organisms (Eisler, 2000). Effects to algal growth have been observed at lead concentrations ranging from 100 to 200,000 µg/L. Clinical signs of lead toxicity in plants include the deformation and disintegration of algae cells and a shortened exponential growth phase. Other effects of lead include a blocking of the pathways that lead to pigment synthesis, thus affecting photosynthesis, cell cycle and division, and ultimately resulting in death. The toxicity of lead to macrophyte growth has been studied using *Spirodela polyrhiza*, *Azolla pinnata*, and *Lemna gibba*. Test durations ranged from 4 to 25 days and test concentrations ranged between 49.7 and 500,000 µg/L.

Waterborne lead is highly toxic to aquatic organisms, with toxicity varying with the species and life stage tested, duration of exposure, form of lead tested, and water quality characteristics. Among the species tested, aquatic invertebrates, such as amphipods and water fleas, were the most sensitive to the effects of lead, with adverse effects being reported at concentrations ranging from 0.45 to 8000 µg/L. Freshwater fish demonstrated adverse effects at concentrations ranging from 10 to >5400 µg/L, depending generally upon water quality parameters. Amphibians tend to be relatively tolerant of lead; however, they may exhibit decreased enzyme activity (e.g., ALAD reduction) and changes in behavior (e.g., hypoxia response behavior). Lead tends to be more toxic with longer-term exposures.

8.2.6 Effects of Lead on Natural Aquatic Ecosystems

8.2.6.1 Introduction

This section discusses the effects of lead on natural aquatic ecosystems. Such effects include changes in species composition and richness, ecosystem function, and energy flow due to lead stress. The format of this section generally follows a conceptual framework for discussing the effects of a stressor such as lead on an ecosystem. This conceptual framework was developed by the EPA Science Advisory Board (Young and Sanzone, 2002). The essential attributes used to describe ecological condition include: landscape condition, biotic condition, chemical and physical characteristics, ecological processes, hydrology and geomorphology and natural disturbance regimes. The majority of the published literature pertaining to lead and aquatic ecosystems focuses on the biotic condition, one of several essential attributes of an ecosystem as described in Young and Sanzone (2002). For the biotic condition, the SAB framework identifies community extent, community composition, trophic structure, community dynamics, and physical structure as factors for assessing ecosystem health. Other factors for assessing the biotic condition such as effects of lead on species, populations, and organism conditions (e.g., physiological status) were discussed in Sections 8.2.4 and 8.2.5.

For natural aquatic ecosystems, the focus of study in the general literature has been on evaluating ecological stress where the sources of lead were from urban and mining effluents (Poulton et al., 1995; Deacon et al., 2001; Mucha et al., 2003). The

statistical methods used when evaluating the effects of lead on aquatic ecosystems are important, as more than one variable may be related to the observed effect. Studied variables include water hardness, pH, temperature, and physical factors such as embeddedness, dominant substrate, and velocity. In most cases single variable statistical techniques were used to evaluate the data. However, in other cases multivariate techniques were used. Therefore, where appropriate, some detail on the statistical methods used is presented.

Although most of the available studies discussed in this section focus on the biotic condition, one case study examining multiple components of the EPA conceptual framework is also included. The remainder of this section describes the effects of lead on the biotic condition.

8.2.6.2 Case Study: Coeur d'Alene River Watershed

The Coeur d'Alene River watershed is an area of Idaho impacted by lead and other metals from historic mining waste releases. Maret et al. (2003) examined several ecological components to determine any negative associations with metals and the watershed communities. The variables examined and associated ecological conditions are presented in Table 8-2.6.1. In addition to measurements of non-metal variables (e.g., dissolved oxygen levels, water temperature and pH, embeddedness), Cd, Pb, and Zn levels were also compared in affected sites versus reference sites.

Some of the above non-metal variables are important to macroinvertebrate communities. For example, a stream with highly embedded substrate can have a lower number of individuals within a species or a different species composition compared to a stream with less embeddedness (Waters, 1995). Macroinvertebrates from the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (EPT) group inhabit the surface of cobble and the interstitial spaces between and underneath cobble. When substrate is embedded, these interstitial spaces are filled, leaving less habitat space for EPT taxa. In another example, water temperature is important; some macroinvertebrates (e.g., stoneflies) are usually only found in cooler water (Harper and Stewart, 1984).

Table 8-2.6.1. Ecological Attributed Studies by Maret et al. (2003) in the Coeur d'Alene Watershed

Ecological Attribute	Subcategory	Measure
Landscape condition	Areal extent landscape pattern	Basin area (km^2) Production mine density/km^2
Biotic Condition	Organism condition population structure/dynamics	Caddisfly tissue concentrations (mg/kg) Number of EPT taxa Density of EPT individuals (no./m^2)
Chemical/physical characteristics	Chemical/physical parameters	Dissolved oxygen (mg/L) Specific conductance (µS/cm) Water temperature (E °C) pH Water hardness (mg/L) Total NO_3 (mg/L) Total P (mg/L) Dissolved NH^3 (mg/L) Sediment Cd, Pb, Zn (mg/kg) Dissolved Cd, Pb, Zn in water (mg/L)
Ecological processes	—	None measured
Hydrology/geomorphology	Channel morphology and distribution	Site elevation (m) Stream gradient (%) Stream discharge (m^3/s) Stream width (m) Stream depth (m) Open canopy (%) Stream velocity (m/s) Embeddedness (%) Dominant substrate (mm)
Natural disturbance regimes	—	None measured

Of the variables examined only metal concentrations, mine density, site elevation, and water temperature were significantly different between reference and mine-affected sites. A Mann-Whitney t-test was used to evaluate statistical differences between reference and test sites for physical and water quality parameters, while Spearman's rank correlation matrices were used to compare all possible response and explanatory variables. Lead concentrations were significantly correlated with the number of mines in

proximity to the watershed. Lead concentrations in sediment and water were strongly correlated to lead levels in whole caddisflies, $r^2 = 0.90$ and 0.63, respectively. Furthermore, mine density was significantly correlated to lead in tissue, $r^2 = 0.64$. Although temperature was significantly different between reference and mine-affected sites, temperature conditions were concluded to be non-limiting to aquatic life. For example, reference and mine-affected sites had at least 15 and 13 obligate cold-water taxa, respectively.

A significant negative correlation between lead in the water column (0.5 to 30 μg/L dissolved) and total taxa richness, EPT taxa richness, and the number of metal-sensitive mayfly species was observed. Similar, significant negative correlations were found between sediment lead levels (132 to 6252 μg/g) and the same macroinvertebrate community metrics and caddisfly tissue levels. Negative correlations were also found between Cd and Zn in the water and sediment and the macroinvertebrate community metrics. In an analysis of cumulative toxicity, lead was judged to be the most significant metal in sediment related to the cumulative toxicity measured. This study provided multiple lines of evidence (i.e., mine density, metal concentrations, bioaccumulation in caddisfly tissue and benthic invertebrate assemblage structure) of the negative impacts of mining in the Coeur d'Alene River, suggesting that lead (and other metals) were primary contributors to the effects observed in the Coeur d'Alene River watershed (Maret et al., 2003).

8.2.6.3 Biotic Condition

In an evaluation of the biotic condition, the SAB framework described by Young and Sanzone (2002) identifies community extent, community composition, trophic structure, community dynamics, and physical structure as essential ecological attributes for assessing ecosystem health. The following two sections describes the effects of lead on community composition, community dynamics, and trophic structure. To date, no available studies were located on the effects of lead on physical structure (e.g., change in tree canopy height, ecosystem succession).

8.2.6.3.1 Ecosystems and Communities, Community Composition

To measure community composition, an inventory of the species/taxa found in the ecological system must be conducted. According the SAB framework, useful measures of composition include the total number of species or taxonomic units, their relative abundance, presence and abundance of native and non-native species, and information on the presence and abundance of focal or special interest species (Young and Sanzone, 2002). Focal or special species of interest can be those that play a critical role in ecosystem processes such as flows of materials or energy within complex food-webs (Young and Sanzone, 2002). Community composition as assessed in lead studies has included the following measures.

- Changes in energy flow or nutrient cycling:
  - Increased or decreased respiration or biomass
  - Increased or decreased turnover/cycling of nutrients

- Changes to community structure:
  - Reduced species abundance (i.e., the total number of individuals of a species within a given area or community)
  - Reduced species richness (i.e., the number of different species present in a community)
  - Reduced species diversity (i.e., a measure of both species abundance and species richness)

Investigators have evaluated the effects of lead on aquatic communities through microcosm and mesocosm studies in natural aquatic systems. Field studies in the general literature have focused on natural systems that were affected by metal stress from various anthropogenic sources. In most of those natural systems, the sources evaluated were from direct mining waste inputs, rather than atmospheric deposition, of lead. Studies published since the 1986 Pb AQCD that describe the effects of lead on natural aquatic ecosystems are presented below and summarized in Table 8-2.6.2. Studies included here evaluated the effects of lead on watersheds, landscapes, aquatic ecosystems, aquatic communities, biodiversity, lakes, rivers, streams, estuaries, wetlands, and species interaction.

Aquatic Microcosm Studies

The examination of simulated aquatic ecosystems (i.e., microcosms) provides limited information on the effects of pollutants on natural systems. Microcosm studies typically focus on only a few aspects of the natural system and do not incorporate all of the ecological, chemical, or biological interactions. Nevertheless, a few microcosm studies have been conducted that indicate potential effects of lead on the community structure of aquatic ecosystems. Fernandez-Leborans and Antonio-García (1988) evaluated the effect of lead on a natural community of freshwater protozoans in simulated aquatic ecosystems and found a reduction in the abundance and composition of protozoan species with increasing lead concentrations (0.05 to 1.0 mg/L) compared to controls. Studies with marine protozoan communities in laboratory microcosms indicated that waterborne lead exposure reduced protozoan abundance, biomass, and diversity at concentrations of 0.02 to 1.0 mg/L Pb. (Fernandez-Leborans and Novillo, 1992, 1994).

Austen and McEvoy (1997) studied the effects of lead on an estuarine meiobenthic community (mainly nematodes) in a microcosm setting using sediment samples collected offshore from England. A multivariate analysis of similarities (ANOSIM) test with square root-transformed data was used to evaluate differences between treatments and controls. Lead was found to significantly affect species abundance at 1343 mg/kg dw relative to a control at 56 mg/kg dw, but no significant adverse effects were observed at the highest dose tested, 1580 mg/kg dw. The authors did not explain why the 1580 mg/kg dw dose was not significant while the 1343 mg/kg dw dose was. None of the lead exposures were significantly different than the controls based on separate univariate tests of abundance, richness, and diversity. There were no other confounding metals in the lead tests, as the experiments were with a single metal dose. In one other mesocosm study, the effects of a mixture of metals (Cu, Cd, Pb, Hg, and Zn) on a salt marsh meiofaunal community were evaluated (Millward et al., 2001). After 30 days exposure, significant reductions in copepod, gastropod, and bivalve abundances were observed at the highest lead exposure concentration, 177 mg/kg dw. Ostracods and nematodes were not affected. The authors believed that the response of the meiofauna taxa to metals was in part due to the various feeding strategies in that deposit feeders were most affected.

Natural Aquatic Ecosystem Studies

Lead stress in aquatic ecosystems has also been evaluated in natural communities. Studies examining community-scale endpoints, however, are complex, and interpretation can be confounded by the variability found in natural systems and the presence of multiple stressors. Natural systems frequently contain multiple metals, making it difficult to attribute observed adverse effects to single metals. For example, macroinvertebrate communities have been widely studied with respect to metals contamination and community composition and species richness (Winner et al., 1980; Chadwick et al., 1986; Clements, 1994). In these studies, multiple metals are evaluated and correlations between observed community level effects are ascertained. The results often indicate a correlation between the presence of one or more metals (or total metals) and the negative effects observed. While, correlation may imply a relationship between two variables, it does not imply causation of effects. The following studies suggest an association between lead concentration and an alteration of community structure and function (see summary in Table 8-2.6.2):

Reduced Primary Productivity and Respiration

Jayaraj et al. (1992) examined the effects of lead on primary productivity and respiration in an algal community of two water bodies. Concentrations of lead in water (6 to 80 mg/L) were found to significantly reduce primary productivity and increase respiration. The authors suggested that increased respiration indicated a greater tolerance to or adaptive mechanisms of the resident heterotrophs to cope with lead stress.

Alterations of Community Structure

Deacon et al. (2001) studied a macroinvertebrate community in mine-affected waters of Colorado. Initially, transplanted bryophytes were used to assess whether metals could bioaccumulate at various mine-affected and unaffected sites (Deacon et al., 2001; Mize and Deacon, 2002). Lead was bioaccumulated by the bryophytes, and median tissue concentrations at mine-affected sites (34 to 299 µg/g dw) were higher than at reference sites (2.5 to 14.7 µg/g dw). Lead concentrations in surface water and sediment ranged from <0.001 to 0.02 mg/L and 145 to 850 mg/kg dw (<63 µm fraction),

Table 8-2.6.2. Essential Ecological Attributes for Natural Aquatic Ecosystems Affected by Lead

Category	Species	Condition Measures	Exposure Medium	Location	Exposure Concentrations	Other Metals Present	Reference
Biotic Condition							
Ecosystems and Communities-Community Composition	Protozoan community	Reduced species abundance and diversity	Marine water	Laboratory microcosm	0.02– 0.05 mg/L	N	Fernandez-Leborans and Novillo (1992)
	Protozoan community	Reduced species abundance	Freshwater water	Laboratory microcosm	0.05–1 mg/L	N	Fernandez-Leborans and Antonio-García (1988)
	Protist community	Reduced species abundance and diversity	Marine water	Laboratory microcosm	1 mg/L	N	Fernandez-Leborans and Novillo (1994)
	Meiofauna community	Reduced abundance	Marine sediment	Laboratory microcosm	177 mg/kg dw	Y	Millward et al. (2001)
	Algal community	Increased respiration	Freshwater	Domestic water stabilization pond	25–80 mg/L	?	Jayaraj et al. (1992)
	Algal community	Decreased primary productivity	Freshwater	Sharana Basaveshwara Tank, India	6–32 mg/L	?	Jayaraj et al. (1992)
	Meiobenthic community (primarily nematodes)	Reduced species abundance No effect on abundance	Marine sediment	Laboratory microcosm	1343 mg/kg dw 1580 mg/kg dw	N	Austen and McEvoy (1997)

Table 8-2.6.2 (cont'd). Essential Ecological Attributes for Natural Aquatic Ecosystems Affected by Lead

Category	Species	Condition Measures	Exposure Medium	Location	Exposure Concentrations	Other Metals Present	Reference
	Macroinvertebrate community	Lower total abundance, decreased taxa, and EPT richness, larger percentage of tolerant species of benthic macroinvertebrates.	Freshwater and sediment	Mining sites in the Upper Colorado Basin	<0.001–0.02 mg/L 145-850 mg/kg dw (<63 µM fraction)	Y	Deacon et al. (2001); Mize and Deacon, (2002)
	Macroinvertebrate community	Negatively correlated with species richness and diversity indices	Estuary sediment	Douro Estuary, Portugal	0.25–192 mg/kg dw	Y	Mucha et al. (2003)
	Macroinvertebrate community	Reduced species abundance	Freshwater sediment	River Ill and tributaries, France	1–16 mg/kg dw	Y	Rosso et al. (1994)
	Fish, crustacean and macroinvertebrate community	Correlation with changes in species abundance and distribution	Marine Sediment	Spencer Gulf, South Australia	156–5270 mg/kg dw	Y	Ward and Young (1982); Ward and Hutchings (1996)
	Chironomid community	Reduced chironomid richness	Whole organism residue	New Brunswick, Canada	40.3–1,387 mg/kg dw (periphyton) 1.6–131 mg/kg dw (chironomid tissue)	Y	Swansburg et al. (2002)
	Macroinvertebrate community	Lead in tissues negatively correlated with taxa richness, EPT richness, chironomid richness, and species density.	Whole organism residue	Clark Fork River, MT	32.2–67.1 mg/kg dw	Y	Poulton et al. (1995)
	Macroinvertebrate community	Lead in tissues negatively correlated with EPT richness and abundance.	Biofilm residues	Boulder River, MT	32–1540 mg/kg dw	Y	Rhea et al. (2004)

Table 8-2.6.2 (cont'd). Essential Ecological Attributes for Natural Aquatic Ecosystems Affected by Lead

Category	Species	Condition Measures	Exposure Medium	Location	Exposure Concentrations	Other Metals Present	Reference
	Macroinvertebrate Community	Lead in tissues and sediment not correlated to diversity and richness	Sediment and whole organism residue	Aquashicola Creek tributaries, Palmerton, PA	7.5–59.5 mg/kg dw (sediment) 0.25-6.03 mg/kg dw (macroinvertebrates)	Y	Carline and Jobsis (1993)
	Fish Community	Lead in tissues and sediment not correlated to diversity and richness	Sediment and whole organism residue	Aquashicola Creek tributaries, Palmerton, PA	7.5–59.5 mg/kg dw (sediment) 0.1-0.86 mg/kg dw (fish)	Y	Carline and Jobsis (1993)
Ecosystems and Communities-Community Dynamics and Trophic Structure	Snails and tadpoles	Lead affected predator-prey interactions	Sediment	Outdoor mini-ecosystems	Not cited	Y	Lefcort et al. (1999)
	Snails and caddisflies	No avoidance of predator by snail. Caddisfly did respond to predator	Water	Field microcosm for snail; in-stream disturbance for caddisfly	27.7–277.6 mg/kg dw (snail tissue) 223–13,507 mg/kg dw (caddisfly tissue)	Y	Lefcort et al. (2000)
	Fathead minnow	Feeding behavior altered	Water	Laboratory microcosm	0.5–1.0 mg/L	N	Weber (1996)
	American toad	No avoidance of lead	Water	Laboratory microcosm	0.5–1.0 mg/L	N	Steele et al. (1991)
	Mummichog	Feeding behavior altered and predator avoidance affected	Water	Laboratory	0.3–1.0 mg/L	N	Weis and Weis (1998)

respectively. The same sites were also evaluated for the effects of various metals on macroinvertebrate communities. Values of total abundance, taxa richness, mayfly, and stonefly abundance were reduced at mining sites. Lead levels along with Cd, Cu, and Zn were correlated with reduced abundance and diversity indices.

Macrobenthic communities studied in an estuary off Portugal were affected by lead at a range from 0.25 to 192 mg/kg dw (Mucha et al., 2003). Species richness was decreased in areas with increased lead concentrations in the sediment. Interpretation of lead effects was complicated by other non-metal stressors, namely sediment particle size and organic matter content. Furthermore, other metals were present (e.g., Al, Cu, Cr, Mn, Zn) and may have affected the community (Mucha et al., 2003).

The effects of lead on oligochaetes in the Ill River and its tributaries in France were evaluated by Rosso et al. (1994). Lead in sediment (5 to 16 μg/g dw at affected sites) was positively correlated to the abundance of the oligochaete, *Nais* sp., and negatively correlated to Tubificidae abundance. Lead was the only metal that was positively correlated to *Nais* species, while other metals were negatively correlated to Tubificidae (Rosso et al., 1994).

The effects of metals and particle size on structuring epibenthic sea grass fauna (fish, mollusks, crustaceans, and polychaetes) was evaluated near a lead smelter in South Australia (Ward and Young, 1982; Ward and Hutchings, 1996). Effluent from the smelter was the primary source of lead and other metal contamination. Species richness and composition were evaluated near the lead smelter along with metal concentrations in sediment. Lead levels in sediment (up to 5270 mg/kg dw) correlated to the negative effects on species richness and composition, while the other metals evaluated had similar correlations. Therefore, lead alone could not be identified as the sole metal causing stress.

Tissue Bioaccumulation Associated with Alterations of Community Structure

Several studies have examined the bioaccumulation of lead in aquatic systems with indices of community structure and function. A focused study on changes in Chironomidae community composition in relation to metal mines (New Brunswick, Canada) identified changes in Chironomidae richness (Swansburg et al., 2002). Lead was not detected (detection limit not given for any matrix) in the water column at any site. However, lead levels in periphyton were significantly higher at mining sites (40.3 to 1387 mg/kg dw) compared to reference sites (not

detected [ND], 33.3 mg/kg dw). Furthermore, lead in chironomids was significantly higher at mine-affected sites (1.6 to 131 mg/kg dw) compared to reference sites (ND,10.2 mg/kg dw). The concentrations in biota indicate that lead is mobile and available to the aquatic community even though water concentrations were undetectable. Chironomidae richness was reduced at the sites receiving mining effluent containing Pb, Cd, Cu, and Zn.

In another study, macroinvertebrate lead tissue concentrations (32.2 to 67.1 mg/kg dw at affected sites) collected from the Clark Fork River, Montana correlated negatively with total richness, EPT richness, and density (Poulton et al., 1995). Mean lead levels were as high as 67.1 mg/kg dw at sites most affected by lead. However, other metals, including Cd, Cu, and Zn, also were negatively correlated with total richness and EPT richness. Therefore, attribution of the observed effects to lead is difficult, as other metals may be contributing factors.

In Montana, the potential effects of metals on macroinvertebrate communities in the Boulder River watershed were evaluated (Rhea et al., 2004). Similar to the approach taken by Poulton et al. (1995), the effects on richness and abundance of EPT taxa were compared to metal concentrations in tissue (i.e., biofilm and macroinvertebrates). Lead levels in biofilm (32 to 1540 mg/kg dw) were significantly correlated with habitat scores and macroinvertebrate indices (e.g., EPT taxa). However, macroinvertebrate tissue lead levels were not significantly correlated with macroinvertebrate community level metrics. As with most natural systems with potential mine impacts, other metals also correlated with community level effects. However, the authors indicated that lead concentrations in biofilm appeared to have the most significant impact on macroinvertebrate metrics.

A detailed investigation of sediment, macroinvertebrates, and fish was conducted for tributaries in the Aquashicola Creek watershed near a former zinc smelter in Palmerton, PA (Carline and Jobsis, 1993). The smelter deposited large amounts of Cd, Cu, Pb, and Zn on the surrounding landscape during its operation from 1898 to 1980. The goal of the study was to evaluate if there was a trend in the metal levels in sediment, macroinvertebrate and fish tissue, and community indices going away from the smelter. Sites were chosen, from 7.8 to 24.6 km from the smelter. There were no clear associations between proximity to the smelter and lead levels in sediment, macroinvertebrate tissue, and fish tissue. Furthermore, there were no associations between proximity to the smelter and macroinvertebrate and fish diversity and

richness. The authors suggested that the transport of metals in the watershed has decreased since the smelter ceased operation, and thereby no effects were observed.

8.2.6.3.2 Ecosystems and Communities, Community Dynamics, and Trophic Structure

As described in the SAB framework, community dynamics include interspecies interactions such as competition, predation, and succession (Young and Sanzone, 2002). Measures of biotic interactions (e.g., levels of seed dispersal, prevalence of disease in populations of focal species) provide important information about community condition. If the community dynamics are disrupted, then the trophic structure may also be disrupted. According to the SAB framework, trophic structure refers to the distribution of species/taxa and functional groups across trophic levels. Measures of trophic structure include food web complexity and the presence/absence of top predators or dominant herbivores. Therefore, this section discusses how aquatic species interactions can be affected by lead. Examples of species interactions can include:

- Predator-prey interactions (e.g., reduced avoidance of predators)
- Prey consumption rate (e.g., increase or decrease in feeding)
- Species competition (e.g., interference with another species, increased aggressive behavior)
- Species tolerance/sensitivity (e.g., the emergence of a dominant species due to contaminant tolerance or sensitivity)

Species interactions are relevant to a discussion about the effects of lead on natural aquatic ecosystems, because effects on species interactions could potentially affect ecosystem function and diversity. Some examples of lead induced changes in species interactions are presented below (see summary in Table 8-2.6.2).

Predator-Prey Interactions

Lefcort et al. (1999) examined the competitive and predator avoidance behaviors of snails and tadpoles in outdoor mini-ecosystems with sediment from a metals-contaminated Superfund site (i.e., Pb, Zn, Cd). Previous investigations of aquatic invertebrates and vertebrates yielded lead tissue concentrations of 9 to 3800 mg/kg dw and 0.3 to 55 mg/kg dw, respectively. Several species interactions were studied in the presence of metal-contaminated sediment:

Snails and tadpoles have similar dietary behaviors. Thus, when placed in the same habitat they will compete for the same food items and negatively affect one another. However, when tadpoles exposed to a predator (i.e., through biweekly additions of 20 mL water from tanks housing sunfish—10 mL from sunfish-fed snails, 10 mL from sunfish-fed tadpoles) were placed with snails, the tadpoles reduced sediment ingestion, while snails increased ingestion. Thus, snails were exposed to greater quantities of metals in sediment.

In an uncontaminated environment, snail recruitment (i.e., reproduction) was reduced in the presence of tadpoles. The addition of tadpoles increased the competition for food in the form of floating algae and the snails switched to feeding on algae that grew on the sediment. This decrease was due to competition alone. The effects on snail recruitment were even higher when tadpoles, the influence of a predator (i.e., sunfish extract), and metals in the sediment were all present. However, the predator effect was indirect in that the tadpoles hid in the algae mats forcing the snails to feed primarily on the benthic algae that grew on the sediment with high metal levels. Furthermore, although not significant, lead levels in snails were higher when tadpoles and sunfish extract were present than when only metals in the sediment were present.

Finally, snail predator avoidance was assessed. Snails (control and lead-exposed) were stimulated with a predator indicator (i.e., crushed snails and an extract of crushed snail). Control snails changed behaviors in the presence of the predator indicator, while exposed snails did not alter their behavior. The authors suggested that metal exposure caused behavioral changes that alter competitive interactions and the perception of predators by the snails. Thus, lead may affect the predator avoidance response of snails.

In further study, Lefcort et al. (2000) examined the predator avoidance behaviors of snails and caddisflies. In separate experiments, the avoidance behavior of the snail, *Physella columbiana*, and four caddisfly genera (*Agrypnia, Hydropsyche, Arctopsyche, Neothremma*) were evaluated. The snails were collected from reference lakes and lakes downstream of the Bunker Hill Superfund site. The snails from the affected lakes generally had higher cadmium, lead, and zinc tissue levels implying previous exposure to these metals. Snail predator avoidance behavior was tested by exposure to crushed snail extract. Snails from the affected lakes did not reduce their activity when exposed to the snail extract, implying a reduced predator avoidance. The lack of response may make the snails at the affected lakes more prone to predation.

The caddisflies were evaluated at 36 sites from six different streams. As with the snails, the caddisflies from the affected streams had higher cadmium, lead and zinc tissue levels. The time for caddisfly larvae to respond (i.e., how long immobile) to disturbance (i.e., lifted from water for 3 seconds and moved to a new location) was evaluated. There was no correlation between tissue metal level and any response variable (Lefcort et al., 2000). Therefore, the authors concluded that preexposure to metals did not reduce predator avoidance for caddisflies.

Weber (1996) examined juvenile fathead minnows exposed to 0, 0.5, or 1.0 ppm lead in water during a 2-week preexposure and 2-week testing period (4 weeks total exposure). Feeding behavior was evaluated by presenting two prey sizes (2-day-old and 7-day-old *Daphnia magna*). Control fish began switching from larger, more difficult-to-capture 7-day-old daphnids to smaller, easier-to-catch 2-day-old prey by day 3. Lead-exposed fish displayed significant switching at day 3 (at 0.5 ppm) or day 10 (at 1.0 ppm). Thus, exposure to lead delayed the altering of prey size choices to less energetically costly prey.

Lefcort et al. (1998) exposed spotted frogs (*Rana luteiventris*) to 0.05 to 50 ppm Pb in water for 3 weeks. High levels of lead reduced the fright response of tadpoles; suggesting a reduced avoidance of predators.

Bullfrog larvae exposed to lead in water (0.78 mg/L) and high or low dissolved oxygen were monitored for respiratory surfacing behavior (Rice et al., 1999). Larvae had a significantly increased number of trips to the water surface regardless of oxygen content. Thus, the authors suggest that lead may affect oxygen uptake such that larvae are under greater predation pressure due to increased time spent at the surface.

Weis and Weis (1998) evaluated the effect of lead exposure on mummichog (*Fundulus heteroclitus*) larvae prey capture rate, swimming behavior, and predator avoidance. Prey capture rates were affected after 4 weeks exposure at 1.0 mg/L lead. The larvae were also more vulnerable to predation by grass shrimp (*Palaemonetes pugio*) at 1.0 mg/L lead. Finally, the swimming behavior of mummichog larvae was affected at 0.3 and 1.0 mg/L lead. Once the larvae were no longer exposed to lead, they recovered their ability to capture prey and avoid predators.

Clearly, exposure to lead does affect the predator-prey interactions and the ability of prey to avoid predators. The effect of lead on these ecological functions may alter community dynamics.

8.2.6.4 Summary

The effects of lead have primarily been studied in instances of point source pollution rather than area-wide atmospheric deposition; thus, the effects of atmospheric lead on ecological condition remains to be defined. The evaluation of point source lead within the EPA Ecological Condition Framework has been examined primarily in relation to biotic conditions. The available literature focuses on studies describing the effects of lead in natural aquatic ecosystems with regard to community composition and species interactions. The effects of lead on the biotic condition of natural aquatic systems can be summarized as follows: there is a paucity of data in the general literature that explores the effects of lead in conjunction with all or several of the various components of ecological condition as defined by the EPA. However, numerous studies are available associating the presence of lead with effects on biotic conditions.

In simulated microcosms or natural systems, environmental exposure to lead in water and sediment has been shown to affect energy flow and nutrient cycling and benthic community structure. In field studies, lead contamination has been shown to significantly alter the aquatic environment through bioaccumulation and alterations of community structure and function. Exposure to lead in laboratory studies and simulated ecosystems may alter species competitive behaviors, predator-prey interactions, and contaminant avoidance behaviors. Alteration of these interactions may have negative effects on species abundance and community structure. In natural aquatic ecosystems, lead is often found coexisting with other metals and other stressors. Thus, understanding the effects of lead in natural systems is challenging given that observed effects may be due to cumulative toxicity from multiple stressors.

8.3 CRITICAL LOADS FOR LEAD IN TERRESTRIAL AND AQUATIC ECOSYSTEMS

8.3.1 Introduction

This section defines critical loads, describes various concepts and methods that are related to the estimation of critical loads, and provides a review of the relevant literature on critical loads.

8.3.1.1 Definitions

Critical loads are defined in a variety of ways depending on the chemicals and endpoints of concern (Pačes, 1998; Skeffington, 1999; U.S. Environmental Protection Agency, 2004a). For the purposes of this section, critical loads are defined as threshold deposition rates of air pollutants that current knowledge indicates will not cause long-term adverse effects to ecosystem structure and function. A critical load is related to an ecosystem's sensitivity to anthropological inputs of a specific chemical. If future inputs of a chemical exceed the critical load for an ecosystem, the chemical is expected to reach or persist at potentially toxic levels in the future. A critical load indicates a potential for future impacts only; a current exceedance of a critical load does not specify whether the current deposition rate of a chemical presents a hazard to the ecosystem.

In order to determine a critical load, the lowest concentration in the receiving medium that poses a potential hazard to a defined ecosystem must first be determined. This concentration, known in the critical loads literature as the critical limit (De Vries et al., 2004), is equal to the effects-based criteria for the most sensitive endpoint in the ecosystem. The critical limit indicates the current potential for adverse effects to an ecosystem.

In contrast to a critical load, a stand-still load is the highest deposition rate of a chemical that will not result in future increases of its concentrations in the environmental media, regardless of the potential for adverse effects at those concentrations. Stand-still loads are also called "acceptable loads" or critical loads calculated using a "stand-still" approach (De Vries et al., 2004) and should not be confused with effects-based critical loads.

8.3.1.2 Historical Perspective

In the 1960s, scientists demonstrated that sulfur emissions on the European continent were contributing to the acidification of Scandinavian lakes. During the 1970s, evidence mounted that air pollutants could travel thousands of miles before deposition occurred, implying that international cooperation was necessary to control acidification. To this end, the European Community (EC) and 34 governments signed the *Convention on Long-range Transboundary of Air Pollution* (CLRTAP) in 1979 under the auspices of the United Nations Economic Commission for Europe (United Nations Economic Commission for Europe (UNECE), 2004).

CLRTAP has since been extended to include eight protocols that regulate air pollutants such as sulfur, nitrogen oxides, heavy metals, persistent organic pollutants, volatile organic compounds, and ozone. In 1988, CLRTAP adopted the critical-load concept, making it basic to the future development of international agreements concerning limitation of the emissions of air pollutants. In 1991, The Coordination Center for Effects (CCE) issued a Technical Report entitled "Mapping Critical Loads for Europe" which presented the first maps of critical loads that were produced as part of the work conducted under the UNECE. Each individual country created maps detailing critical loads and levels of acidity within its boundaries. The maps were then used by CCE to create a Europe-wide map of critical loads (Hettelingh et al., 1991) that is used in combination with air emissions and deposition data to guide negotiations between nations and reduce the gap between critical loads and deposition (Skeffington, 1999). The first international agreement on pollution control based on critical loads was the second Sulfur Protocol, which was established in Oslo (United Nations Economic Commission for Europe (UNECE), 1994) within CLRTAP.

Since 1991, CCE has issued biennial technical status reports on critical loads and critical thresholds of acidification, eutrophication, sulfur, nitrogen, and nitrogen oxide (Coordination Center for Effects (CCE), 2005). Progress on data and methodologies is reviewed annually in CCE Mapping workshops. Recent CCE reports focus on scientific and technical support for the revision of protocols as well as time horizons for recovery from ecosystem damage.

Many of the signatory governments to CLRTAP have adopted the critical load concept for determining national emission control polices. Canada has also committed to a critical load approach for controlling acid deposition. In 1998, federal, provincial, and territorial Energy and Environment Ministers signed *The Canada-wide Acid Rain Strategy for Post-2000*. According to Environment Canada, the primary long-term goal of the *Strategy* is to achieve critical loads (or the threshold level) for acid deposition across Canada (Environment Canada, 2003).

The Ministry of Environment in the Netherlands took the initiative to develop analogous methods for the calculation of critical loads for heavy metals, methods that would be valid in the context of CLPTRP (De Vries et al., 2004). Beginning in the mid-1990s, these methods were developed through a series of manuals, international workshops, and expert meetings (De Vries et al., 2004). Participating nations completed a voluntary preliminary critical load mapping exercise for Pb and cadmium in Europe in 2002 (Hettelingh et al., 2002).

The EPA Pb AQCD, Volume II (U.S. Environmental Protection Agency, 1986a) largely predates the development of the concept of critical loads, and does not include this topic. The EPA 2004 Air Quality Criteria for Particulate Matter, Volume I (U.S. Environmental Protection Agency, 2004a) include a brief discussion of the key elements of the critical loads framework general to any air pollutant. To date, the critical loads framework has not been used for regulatory purposes in the United States for any chemical.

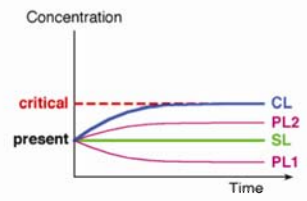
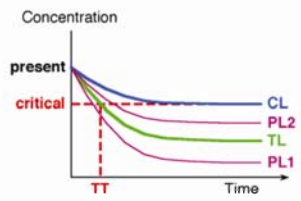
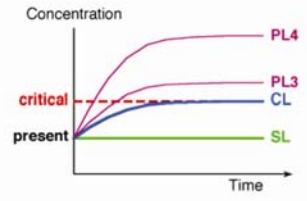
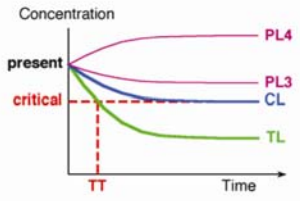
8.3.2 Application of Critical Loads to Terrestrial and Aquatic Ecosystems

A combinatorial application of critical limit and critical load allows one to assess current risk while simultaneously estimating future risk from exposure to a chemical (De Vries et al., 2004). Figure 8-3.1 shows that four combinations of critical load and limit exceedance or non-exceedance are possible for a given ecosystem (Figure 1 of De Vries et al. [2004]). For example, if a current risk is indicated by an exceedance of the critical limit for Pb due to historical Pb deposition, but current inputs of Pb to the ecosystem are below the critical load (lower left corner), the critical load model predicts that Pb concentrations will fall below the critical limit at some point in the future if Pb deposition is maintained at the present level. If current soil concentrations are below the critical limit (upper right corner), inputs greater than the critical load will not result in exceedance of the critical limit for some period of time, but continued exceedance of a critical load will eventually lead to an exceedance of the critical limit.

The time until a critical limit is exceeded (critical time) can also be predicted using the critical load model (Pačes, 1998). This requires knowledge of current concentrations, the critical load, and predicted deposition rates. Critical times may be useful for setting priorities between ecosystems with critical load exceedances or between different chemicals.

8.3.3 Calculation of Critical Loads

This section summarizes the various methods used to calculate critical loads (De Vries et al., 2001, 2002, 2004; Groenenberg et al., 2002), with an emphasis on the most recent material.

	No critical limit exceedance	Critical limit exceedance
No critical load exceedance	*No damage at present or foreseen:*  ➔ Keep the Present Load (more stringent than Critical Load)	*Present damage but recovery in progress:*  ➔ Keep the Present Load (more stringent than Critical Load) or ➔ Consider Target Load to reach the critical limit in a defined time period (more stringent than Critical Load)
Critical load exceedance	*Future damage foreseen:*  ➔ Consider Critical Load (emissions must decrease, even if concentrations in the ecosystem are allowed to increase further at critical load)	*Present damage, no recovery foreseen:*  ➔ Consider Critical Load (decrease of concentrations in the ecosystem down to critical limit in the long term) or ➔ Consider Target Load to reach the critical limit in a defined time period (more stringent than Critical Load)

CL - Critical load; PL - present load (2 cases); SL - Stand-still load; TL - Target load; TT - Target time

Figure 8-3.1. The predicted development of metal concentrations in ecosystems for four cases of exceedance or non-exceedance of critical limits and of critical loads of heavy metals, respectively.

Source: Taken from DeVries et al. (2004).

8.3.3.1 Critical Limits

To determine the critical limit, effects-based criteria for the major ecological endpoints should be developed for the ecosystem of concern. Criteria may be developed for any receptor that is exposed to the chemical of concern deposited in the ecosystem. In terrestrial ecosystems,

possible ecological endpoints include effects from direct contact of invertebrates or plants with soil and ingestion of plants by herbivores. Effects-based criteria for use in defining the critical limit should be derived from ecotoxicological data appropriate to the most sensitive endpoint (De Vries et al., 2004). Regardless of the selected endpoint, the critical limit should be defined as a concentration in the medium that receives the depositional load, typically soil in terrestrial ecosystems and surface water in aquatic ecosystems. To derive these values, uptake and/or food chain modeling may be necessary.

Many critical load calculations rely on ecological effects criteria developed by government agencies in individual countries (Pačes, 1998; De Vries et al., 1998; Van Den Hout et al., 1999; Skjelkvåle et al., 2001). Criteria for Pb vary widely and can be the largest source of uncertainty in a critical load calculation (Van Den Hout et al., 1999). One reason for the wide range in estimates of effects criteria is that Pb speciation is often not taken into account. This can result in variation in estimates of concentration for total Pb that is associated with adverse effects, since the fraction of Pb available to cause a toxic effect depends on chemical factors such as the pH or organic matter content (Lofts et al., 2004). To develop effects-based criteria that are applicable to media with a pH or organic matter content different from the test medium, it is more appropriate to develop criteria based on the free concentration of Pb rather than the total concentration of Pb.

8.3.3.2 Models

Critical loads for heavy metals are typically calculated using a steady state model that ignores internal metal cycling and keeps the calculations as simple as possible (De Vries et al., 2004). The critical load is equal to the atmospheric input flux, which equals the sum of the output fluxes from the system minus the other input fluxes (e.g., weathering) when the concentration of Pb is at the critical limit. The input flux of heavy metals via weathering is sometimes neglected, because quantitative estimates are highly uncertain, and weathering is generally thought to be a relatively minor process (De Vries et al., 2004).

More complex methods may be used to calculate critical loads. For example, dynamic models can be used to model the change of concentrations in soil or water over time (Pačes, 1998). These models are most valuable when the time to steady state is very long compared to the time of interest. Using these models, the critical load is the deposition rate that leads to

concentrations equal to the critical limit as the model approaches steady state. Fate and transport models that include internal cycling can also be used in place of simple mass balance models (Doyle et al., 2003) that may improve the accuracy of the models.

Terrestrial Model

If internal cycling and weathering of Pb is neglected and atmospheric deposition is the only important source of Pb to the system, the critical load in a terrestrial ecosystem is equal to the sum of the most important fluxes out of the system, leaching, and uptake by harvested plants:

$$CL(Pb) = Pb_u + Pb_{le(crit)} \quad (8\text{-}5)$$

where:

$CL(Pb)$ = critical load of Pb (mass per area-year)
Pb_u = metal net uptake in harvestable parts of plants at the critical limit (mass per area-year)
$Pb_{le(crit)}$ = leaching flux of Pb (dissolved and particulate) from the soil layer at the critical limit (mass per area-year)

When applying a mass balance model, it is important to define the boundaries of the compartment such that all significant fluxes in and out of the compartment can be accounted for.

Uptake of Pb by harvested vegetation may be an important flux out of agricultural soil or forested soil that is actively logged. In ecosystems that are not harvested, the steady state model assumes that uptake by plants is balanced by deposition of Pb from decaying vegetation.

The flux out of the system due to uptake in harvested plants is calculated as follows:

$$Pb_u = f_{Pb,u,z} * Y_{ha} * [Pb]_{ha} \quad (8\text{-}6)$$

where:

$f_{Pb,u,z}$ = fraction of net Pb uptake from soil within the considered layer (dimensionless)
Y_{ha} = annual yield of harvestable biomass (mass per area-year)
$[Pb]_{ha}$ = metal concentration of harvestable parts of plants (Pb per unit mass)

The net fraction of metal uptake from soil within the considered layer corrects for Pb measured in harvested vegetation that is taken up via direct deposition onto the plant or from soil outside of the considered soil layer.

The yield of harvestable biomass should only include the parts of plants that are removed from the system. Tree leaves, stalks remaining after harvest of agricultural land, and other parts that remain in the considered terrestrial ecosystem should not be included in the yield.

De Vries et al. (2004) recommends that data for metal content in harvestable biomass should be taken from unpolluted areas. This leads to more conservative critical loads than using the metal content at the critical load. If the selected endpoint for the critical limit is related to the concentration in harvested plants rather than a concentration in soil, that critical concentration should be used in place of actual metal content in harvestable biomass.

The critical leaching flux from the topsoil can be calculated as follows:

$$M_{cl(crit)} = Q_{le} * [Pb]_{tot,sdw(crit)} \quad (8\text{-}7)$$

where:

Q_{le} = flux of drainage water leaching from the considered soil layer (volume/year)

$[Pb]_{tot,sdw(crit)}$ = critical total concentration of Pb in soil drainage water (mass per volume)

The total concentration of Pb in soil drainage water is the sum of all species of dissolved and particulate Pb that leach out of the system in drainage water. De Vries et al. (2004) suggests that Pb that is sorbed to suspended particulate matter should be neglected so that total Pb is equal to dissolved Pb, as concentrations of suspended solids are difficult to estimate. Dissolved Pb may exist as free ions, organic complexes, or inorganic complexes.

The drainage water flux leaching from the topsoil can be calculated as follows:

$$Q_{le} = P - E_i - E_s - f_{Et,z} * E_t \quad (8\text{-}8)$$

where:

P = Precipitation (volume per area-time)

E_i = Interception evaporation (volume per area-time)

E_s = Soil evaporation within the topsoil (volume per area-time)

$f_{Et,z}$ = Plant transpiration (volume per area-time)

E_t = Fraction of water uptake within the topsoil by roots (unitless)

De Vries et al. (2004) recommends default values for some of these parameters and provides an alternative calculation method for sites with detailed hydrologic data as part of the guidance document.

Aquatic Model

If internal cycling and weathering of Pb is neglected and atmospheric deposition is the only important source of Pb to the system, the critical load in an aquatic ecosystem is equal to the sum of the most important fluxes out of the system, uptake by harvested plants in the catchment, sedimentation, and lateral outflow from the catchment:

$$CL(Pb) = Pb_u + Pb_{sed(crit)} * A_l / A_c + Pb_{loc,crit} \quad (8\text{-}9)$$

where:

Pb_u = removal of Pb by harvesting of vegetation in the catchment (mass per area-time)
$Pb_{sed(crit)}$ = removal of Pb by sedimentation at the critical load (mass per area-time)
$Pb_{loc,crit}$ = lateral Pb outflow from the catchment at the critical load (mass per area-time)
A_l = lake area
A_c = catchment area

It is important to carefully define the boundaries of the aquatic system, so that all inflows and outflows may be fully accounted for. Current guidance recommends including the entire watershed within the system, rather than confining the system to a single lake or stream (De Vries et al., 2004). In stream water, removal of Pb due to sedimentation does not need to be considered, simplifying the equation to the following:

$$CL(Pb) = Pb_u + Pb_{loc,crit} \quad (8\text{-}10)$$

De Vries et al. (2004) recommends that critical loads should be calculated for stream waters only, due to a high level of uncertainty in the rate of removal via sedimentation or other removal mechanisms within a lake. Critical loads for streams are protective of nearby lakes, because the critical loads calculated using this methodology will be lower for streams than for lakes.

Calculation of removal of Pb by harvesting of vegetation in the catchment is similar to that in terrestrial ecosystems, with $f_{Pb,u}$ equal to 1, since the entire catchment is now included.

The critical lateral Pb outflow from the catchment is the product of the lateral outflow flux of water and the total concentration of Pb in the outflow water at the critical limit. The outflow flux of water is calculated from the outflow divided by the catchment area.

8.3.4 Critical Loads in Terrestrial Ecosystems

Critical loads of Pb have been calculated using simple mass balance, dynamic, and probabilistic models for forested and agricultural land in Europe and Canada in a handful of preliminary studies. The methods and model assumptions used to calculate critical loads vary widely between these studies and little attempt has been made to validate the models that were used, so it is not known how much various simplifying assumptions affect the results.

Pačes (1998) used data from a small agricultural catchment in the Czech Republic that is typical of agricultural land in that country to calculate critical loads for Pb and other heavy metals. The critical loads were calculated using a simple dynamic box model. The fluxes into the system included atmospheric deposition, agricultural inputs, and weathering of bedrock and the fluxes out of the system included biological uptake and runoff. The model assumed that inputs of metals to the system are independent of their concentrations in soil but that outputs are proportional to the concentration of biologically active metal. The author defined biologically-active metal as the concentration of metal in soil that can be extracted in a 2 M nitric acid solution. This method was used to set a Czech state norm designed to be protective for soil systems that is used as the critical limit in this study. Using the model, Pačes determined that the critical limit was not presently exceeded, but that the critical load is exceeded. However, the critical time was almost 1,000 years. Therefore, the model predicts that Pb will continue to accumulate in Czech agricultural soil and will eventually pose a potential risk if current inputs continue. The author identified the simplifying assumptions used to calculate fluxes out of the system as the major source of uncertainty.

Van den Hout et al. (1999) calculated critical loads for Pb and other pollutants in the organic and mineral soil layers of forested ecosystems. Atmospheric deposition was assumed to be the only inflow, and outflows from soil were assumed to occur due to biological uptake and leaching. Net heavy metal uptake by the forest was set equal to the rate of water uptake by

vegetation multiplied by the water concentration and a "preference factor" that indicates the preference of the vegetation for the metal relative to water. Water flux was estimated from precipitation, soil evaporation, and transpiration data. An equilibrium speciation model that takes inorganic and organic ligands into account was used to estimate dissolved concentrations of Pb in leachate. Results were strongly dependant on the critical limits that were chosen. Using the most stringent levels, critical loads were exceeded over much of Europe. The time to steady state was estimated to be hundreds of years. Speciation of Pb was identified as an important source of uncertainty.

Reinds et al. (2002) used the guidance prepared by De Vries et al. (2002) to calculate critical loads in the mineral topsoil of forested and agricultural ecosystems across 80,000 areas of the European continent. The median critical load for Pb in Europe was 25 g ha^{-1} $year^{-1}$ using this methodology. The drainage water flux leaching from the topsoil was the dominant term in the model, so critical loads followed the spatial pattern of net runoff (excess precipitation) across Europe.

Probst et al. (2003) calculated critical loads for Pb for forested sites in France. Weathering rates were determined using a model for representative French soil samples. The biomass uptake of Pb was derived using National Forestry Inventory data for the average annual biomass growth and data for the Pb content in biomass. An uptake factor scaled down to the considered depth was applied. Leaching of Pb was calculated using runoff data and dissolved Pb concentrations in soil solution. Critical loads at the French site varied over a wide range (4.9 to 133 g ha- $year^{-1}$). Critical loads were controlled mainly by net runoff. Weathering rates were small compared to leaching and biomass uptake rates.

Doyle et al. (2003) used a probabilistic assessment to calculate critical loads in terrestrial and aquatic (see following section) ecosystems on the Canadian Shield. The terrestrial model used an analytical solution to the convection/dispersion equation. The model only considered soluble metal in the flux to soil and assumed that the insoluble fraction was not available. Metals were assumed to be sorbed onto immobile soil solids according to an equilibrium distribution (Kd) relationship. The input parameters were selected to represent boreal forest and Canadian Shield conditions. Best estimate inputs were used for deterministic evaluation and distributions of values were used in a probabilistic assessment. The model inputs included net water flux, effective water velocity, moisture content of soil, pH, dispersion coefficient, and Kd. The 25th

percentile critical loads (47 mg/m^3 per year for Pb) were compared to current deposition rates to evaluate risk.

In spite of the variation in methods and model assumptions used to calculate critical loads for Pb in the studies discussed above, some general conclusions may be drawn. The critical limit is the most important value for determining the value of the critical load. Wide variations in available effects levels, makes this parameter one of the most important sources of uncertainty when calculating critical loads in terrestrial ecosystems. Spatial variations in critical loads for Pb are largely controlled by net runoff. Weathering and uptake by harvestable vegetation were less important. The time to reach steady state is several hundred years in the two studies that used dynamic models to determine critical loads.

8.3.5 Critical Loads in Aquatic Ecosystems

Doyle et al. (2003) modeled critical loads in surface water bodies assuming complete mixing with dilution water entering from the terrestrial catchment area. Loss of metal was also assumed to occur though downstream flushing and burial in sediment. Transfer of metal to sediment was modeled as a first-order process dependant on the dissolved concentration and pH. The inputs to the model included the following: water body area, terrestrial catchment area, water body depth, sediment accumulation rate, thickness of biologically active sediment, net precipitation, and water pH. The fist-order rate constant for transfer to sediment were correlated to pH. The model reaches steady state within a few years. Transfer of Pb from the terrestrial catchment to the water body was neglected, because the time to steady state could be on the order of 10,000 years if the model included this source of Pb. However, the authors cited a separate calculation that indicated that neglect of transfer of Pb from the catchment may lead to a 5-fold underestimation of Pb concentrations in the surface water.

These results indicate that Pb run-off from soil is more important than direct atmospheric deposition to the surface water bodies considered in this study. Due to the long times required to achieve steady state, the critical load methodology may not be appropriate for Pb in aquatic systems.

8.3.6 Limitations and Uncertainties

The largest sources of uncertainty identified in studies of critical loads for Pb include the following:

- Steady-state assumption
- Derivation of the critical limit
- Lead speciation
- Soil runoff as an input to aquatic ecosystems

The critical load is calculated for steady state conditions, but the time for Pb to reach steady state concentrations can be as long as several centuries. Thus, dynamic models are often used to predict Pb concentrations over shorter time frames. Dynamic modeling requires additional knowledge about current concentrations in the considered ecosystem. For regulatory purposes, use of dynamic modeling requires that a target time be set in order to calculate a critical load.

Criteria for the protection of soil and for the protection of aquatic organisms vary over a wide range from country to country. Use of the critical loads method for international negotiations will require implementation of a consistent calculation methodology that takes into account the effect of Pb speciation on toxicity over a range of soil types and chemical conditions.

Speciation strongly influences the toxicity of Pb in soil and water and partitioning between dissolved and solid phases determines the concentration of Pb in soil drainage water, but it has not been taken into account in most of the critical load calculations for Pb performed to date. Recent guidance for heavy metals has begun to emphasize the importance of speciation to critical load calculations and suggest methods to calculate speciation (De Vries et al., 2004). To this end, Lofts et al., (2004) developed critical limit functions for several metals, including Pb, that take into account the effects of pH, organic matter, and the protective effects of cations on speciation.

Runoff of Pb from soil may be the major source of Pb into aquatic systems. However, little attempt has been made to include this source into critical load calculations for aquatic systems due to the complexity of including this source in the critical load models.

8.3.7 Conclusions

Preliminary efforts to calculate critical loads for Pb in terrestrial and aquatic ecosystems have so far relied on a variety of calculation methods and model assumptions. Efforts to refine and standardize methods for the calculation of critical loads for heavy metals which are valid in the context of CLPTRP are ongoing. At this time, the methods and models commonly used for the calculation of critical loads have not been validated for Pb. Many of the methods neglect the speciation of Pb when estimating critical limits, the uptake of Pb into plants, and the outflux of Pb in drainage water, limiting the utility of current models.

Future efforts should focus on fully incorporating the role of Pb speciation into critical load models, and validating the assumptions used by the models.

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