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Annexes for the Integrated Science Assessment for Oxides of Nitrogen – Health Criteria

(First External Review Draft)

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

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Annexes for the Integrated Science Assessment for Oxides of Nitrogen – Health Criteria

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Annex List of Tables

(cont'd)

Authors, Contributors, and Reviewers

Authors

Dr. Dennis J. Kotchmar (NO_x Team Leader)—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Mary Ross (Branch Chief)—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Kathleen Belanger, Yale University, Epidemiology and Public Health, 60 College Street, New Haven, CT 06510-3210

Dr. James S. Brown—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Douglas Bryant—Cantox Environmental Inc., 1900 Minnesota Court, Mississauga, Ontario L8S 1P5

Dr. Ila Cote—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Mark Frampton—Strong Memorial Hospital, 601 Elmwood Ave., Box 692, Rochester, NY 14642-8692

Dr. Janneane Gent—Yale University, CPPEE, One Church Street, 6th Floor, New Haven, CT 06510

Dr. Vic Hasselblad—Duke University, 29 Autumn Woods Drive, Durham, NC 27713

Dr. Kazuhiko Ito—New York University School of Medicine, 57 Old Forge Road, Tuxedo, NY 10987

Dr. Jee Young Kim—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Ellen Kirrane—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Thomas Luben—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Andrew Maier—Toxicology Excellence for Risk Assessment, 2300 Montana Avenue, Suite 409, Cincinnati, OH 45211

Authors, Contributors, and Reviewers
(cont'd)

Authors
(cont'd)

Dr. Qingyu Meng—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Joseph Pinto—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Paul Reinhart—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. David Svendsgaard—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Lori White—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Contributors

Dr. Dale Allen, University of Maryland, College Park, MD

Dr. Jeffrey Arnold—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Ms. Louise Camalier, U.S. EPA, OAQPS, Research Triangle Park, NC

Dr. Russell Dickerson, University of Maryland, College Park, MD

Dr. Tina Fan, EOHSI/UMDNJ, Piscataway, NJ

Dr. Arlene Fiore, NOAA/GFDL, Princeton, NJ

Dr. Panos Georgopoulos, EOHSI/UMDNJ, Piscataway, NJ

Dr. Larry Horowitz, NOAA/GFDL, Princeton, NJ

Dr. William Keene, University of Virginia, Charlottesville, VA

Dr. Randall Martin, Dalhousie University, Halifax, Nova Scotia

Authors, Contributors, and Reviewers
(cont'd)

Contributors
(cont'd)

Dr. Maria Morandi, University of Texas, Houston, TX

Dr. William Munger, Harvard University, Cambridge, MA

Mr. Charles Piety, University of Maryland, College Park, MD

Dr. Sandy Sillman, University of Michigan, Ann Arbor, MI

Dr. Helen Suh, Harvard University, Boston, MA

Ms. Debra Walsh—National Center for Environmental Assessment (B243-01),
U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Charles Wechsler, EOHSI/UMDNJ, Piscataway, NJ

Dr. Clifford Weisel, EOHSI/UMDNJ, Piscataway, NJ

Dr. Jim Zhang, EOHSI/UMDNJ, Piscataway, NJ

Reviewers

Dr. Tina Bahadori—American Chemistry Council, 1300 Wilson Boulevard, Arlington, VA 22209

Dr. Tim Benner—Office of Science Policy, Office of Research and Development, Washington, DC 20004

Dr. Daniel Costa—National Program Director for Air, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Robert Devlin—National Health and Environmental Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Chapel Hill, NC

Dr. Judy Graham—American Chemistry Council, LRI, 1300 Wilson Boulevard, Arlington, VA 22209

Dr. Stephen Graham—Office of Air and Radiation, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Authors, Contributors, and Reviewers
(cont'd)

Reviewers
(cont'd)

Ms. Beth Hassett-Sipple—U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Dr. Gary Hatch—National Health and Environmental Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Scott Jenkins—Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency (C504-02), Research Triangle Park, NC 27711

Dr. David Kryak—National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Mr. John Langstaff—U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Dr. Morton Lippmann—NYU School of Medicine, 57 Old Forge Road, Tuxedo, NY 10987

Dr. Thomas Long—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Karen Martin—Office of Air and Radiation, U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Dr. William McDonnell—William F. McDonnell Consulting, 1207 Hillview Road, Chapel Hill, NC 27514

Dr. Dave McKee—Office of Air and Radiation/Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Dr. Lucas Neas—National Health and Environmental Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Chapel Hill, NC 27711

Dr. Russell Owen—National Health and Environmental Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Haluk Ozkaynak—National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Authors, Contributors, and Reviewers
(cont'd)

Reviewers
(cont'd)

Dr. Jennifer Peel—Colorado State University, 1681 Campus Delivery, Fort Collins, CO 80523-1681

Mr. Harvey Richmond—Office of Air Quality Planning and Standards/Health and Environmental Impacts Division, U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Mr. Joseph Somers—Office of Transportation and Air Quality, U.S. Environmental Protection Agency, 2000 Traverwood Boulevard, Ann Arbor, MI 48105

Ms. Susan Stone—U.S. Environmental Protection Agency (C504-06), Research Triangle Park, NC 27711

Dr. John Vandenberg—National Center for Environmental Assessment (B243-01), Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Alan Vette—National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Mr. Ron Williams—National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. William Wilson—Office of Research and Development, National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

U.S. Environmental Protection Agency Project Team for Development of Integrated Scientific Assessment for Oxides of Nitrogen

Executive Direction

Dr. Ila Cote (Acting Director)—National Center for Environmental Assessment-RTP Division, (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Scientific Staff

Dr. Dennis Kotchmar (NO_x Team Leader)—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Jeff Arnold—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. James S. Brown—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Jee Young Kim—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Ellen Kirrane—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Tom Long—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Thomas Luben—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Quingyu Meng—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Joseph Pinto—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Paul Reinhart—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Mary Ross—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

U.S. Environmental Protection Agency Project Team for Development of Integrated Scientific Assessment for Oxides of Nitrogen

(cont'd)

Scientific Staff
(cont'd)

Dr. David Svendsgaard—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. Lori White—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Dr. William Wilson—National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Technical Support Staff

Ms. Emily R. Lee—Management Analyst, National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Ms. Christine Searles—Management Analyst, National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Ms. Debra Walsh—Program Analyst, National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Mr. Richard Wilson—Clerk, National Center for Environmental Assessment (B243-01), U.S. Environmental Protection Agency, Research Triangle Park, NC 27711

Document Production Staff

Ms. Barbra H. Schwartz—Task Order Manager, Computer Sciences Corporation, 2803 Slater Road, Suite 220, Morrisville, NC 27560

Mr. John A. Bennett—Technical Information Specialist, Library Associates of Maryland, 11820 Parklawn Drive, Suite 400, Rockville, MD 20852

Mrs. Melissa Cesar—Publication/Graphics Specialist, Computer Sciences Corporation, 2803 Slater Road, Suite 220, Morrisville, NC 27560

U.S. Environmental Protection Agency Project Team for Development of Integrated Scientific Assessment for Oxides of Nitrogen

(cont'd)

Document Production Staff
(cont'd)

Mrs. Rebecca Early—Publication/Graphics Specialist, TekSystems, 1201 Edwards Mill Road, Suite 201, Raleigh, NC 27607

Mr. Eric Ellis—Records Management Technician, InfoPro, Inc., 8200 Greensboro Drive, Suite 1450, McLean, VA 22102

Ms. Stephanie Harper—Publication/Graphics Specialist, TekSystems, 1201 Edwards Mill Road, Suite 201, Raleigh, NC 27607

Ms. Sandra L. Hughey—Technical Information Specialist, Library Associates of Maryland, 11820 Parklawn Drive, Suite 400, Rockville, MD 20852

Dr. Barbara Liljequist—Technical Editor, Computer Sciences Corporation, 2803 Slater Road, Suite 220, Morrisville, NC 27560

Ms. Molly Windsor—Graphic Artist, Computer Sciences Corporation, 2803 Slater Road, Suite 220, Morrisville, NC 27560

U.S. Environmental Protection Agency
Science Advisory Board (SAB)
Staff Office Clean Air Scientific Advisory Committee (CASAC)
CASAC NO_x and SO_x Primary NAAQS Review Panel

Chair

Dr. Rogene Henderson*, Scientist Emeritus, Lovelace Respiratory Research Institute, Albuquerque, NM

Members

Mr. Ed Avol, Professor, Preventive Medicine, Keck School of Medicine, University of Southern California, Los Angeles, CA

Dr. John R. Balmes, Professor, Department of Medicine, Division of Occupational and Environmental Medicine, University of California, San Francisco, CA

Dr. Ellis Cowling*, University Distinguished Professor At-Large, North Carolina State University, Colleges of Natural Resources and Agriculture and Life Sciences, North Carolina State University, Raleigh, NC

Dr. James D. Crapo [M.D.]*, Professor, Department of Medicine, National Jewish Medical and Research Center, Denver, CO

Dr. Douglas Crawford-Brown*, Director, Carolina Environmental Program; Professor, Environmental Sciences and Engineering; and Professor, Public Policy, Department of Environmental Sciences and Engineering, University of North Carolina at Chapel Hill, Chapel Hill, NC

Dr. Terry Gordon, Professor, Environmental Medicine, NYU School of Medicine, Tuxedo, NY

Dr. Dale Hattis, Research Professor, Center for Technology, Environment, and Development, George Perkins Marsh Institute, Clark University, Worcester, MA

Dr. Patrick Kinney, Associate Professor, Department of Environmental Health Sciences, Mailman School of Public Health, Columbia University, New York, NY

Dr. Steven Kleeberger, Professor, Laboratory Chief, Laboratory of Respiratory Biology, NIH/NIEHS, Research Triangle Park, NC

Dr Timothy Larson, Professor, Department of Civil and Environmental Engineering, University of Washington, Seattle, WA

U.S. Environmental Protection Agency
Science Advisory Board (SAB)
Staff Office Clean Air Scientific Advisory Committee (CASAC)
CASAC NO_x and SO_x Primary NAAQS Review Panel
(cont'd)

Members
(cont'd)

Dr. Kent Pinkerton, Professor, Regents of the University of California, Center for Health and the Environment, University of California, Davis, CA

Mr. Richard L. Poirot*, Environmental Analyst, Air Pollution Control Division, Department of Environmental Conservation, Vermont Agency of Natural Resources, Waterbury, VT

Dr. Edward Postlethwait, Professor and Chair, Department of Environmental Health Sciences, School of Public Health, University of Alabama at Birmingham, Birmingham, AL

Dr. Armistead (Ted) Russell*, Georgia Power Distinguished Professor of Environmental Engineering, Environmental Engineering Group, School of Civil and Environmental Engineering, Georgia Institute of Technology, Atlanta, GA

Dr. Richard Schlesinger, Associate Dean, Department of Biology, Dyson College, Pace University, New York, NY

Dr. Christian Seigneur, Vice President, Atmospheric and Environmental Research, Inc., San Ramon, CA

Dr. Elizabeth A. (Lianne) Sheppard, Research Professor, Biostatistics and Environmental & Occupational Health Sciences, Public Health and Community Medicine, University of Washington, Seattle, WA

Dr. Frank Speizer [M.D.]*, Edward Kass Professor of Medicine, Channing Laboratory, Harvard Medical School, Boston, MA

Dr. George Thurston, Associate Professor, Environmental Medicine, NYU School of Medicine, New York University, Tuxedo, NY

Dr. James Ultman, Professor, Chemical Engineering, Bioengineering Program, Pennsylvania State University, University Park, PA

Dr. Ronald Wyzga, Technical Executive, Air Quality Health and Risk, Electric Power Research Institute, P.O. Box 10412, Palo Alto, CA

U.S. Environmental Protection Agency
Science Advisory Board (SAB)
Staff Office Clean Air Scientific Advisory Committee (CASAC)
CASAC NO_x and SO_x Primary NAAQS Review Panel
(cont'd)

SCIENCE ADVISORY BOARD STAFF

Mr. Fred Butterfield, CASAC Designated Federal Officer, 1200 Pennsylvania Avenue, N.W., Washington, DC, 20460, Phone: 202-343-9994, Fax: 202-233-0643 (butterfield.fred@epa.gov) (Physical/Courier/FedEx Address: Fred A. Butterfield, III, EPA Science Advisory Board Staff Office (Mail Code 1400F), Woodies Building, 1025 F Street, N.W., Room 3604, Washington, DC 20004, Telephone: 202-343-9994)

* Members of the statutory Clean Air Scientific Advisory Committee (CASAC) appointed by the EPA Administrator

Annex Abbreviations and Acronyms

α	alpha; probability value
AA	arachidonic acid
ACCENT	European Union project Atmospheric Composition Change: the European Network of Excellence
AgNOR	argyrophilic nucleolar organizer region
AIRPEX	Air Pollution Exposure (model)
AIRQUIS	Air Quality Information System (model)
AIRS	Aerometric Information Retrieval System
AM	alveolar macrophage
AMF	air mass factor
AMT	average medial thickness
APEX	Air Pollution Exposure (model)
APIMS	atmospheric pressure ionization mass spectrometer
AQCD	Air Quality Criteria Document
AQEG	Air Quality Expert Group
ATS	American Thoracic Society
ATTILA	type of Lagrangian model
BAL	bronchoalveolar lavage
BALF	bronchoalveolar lavage fluid
BERLIOZ	Berlin Ozone Experiment
BHPN	*N*-bis(2-hydroxyl-propyl)nitrosamine
BLF	bronchial lavage fluid
BME	Bayesian Maxim Eutropy
Br	bromine
Br^-	bromine ion
Br_2	molecular bromine
BrCl	bromine chloride
BrdU	bromodeoxyuridine
BrO	bromine oxide
C	concentration
C × T	concentration × time; concentration times duration of exposure
CAA	Clean Air Act
CAPs	concentrated ambient particles
CARB	California Air Resources Board
CASAC	Clean Air Scientific Advisory Committee
CB4	Carbon Bond 4 (chemical mechanism)

CC10	Clara cell 10-kDa protein
CCN	cyanomethylidyne radical
CD	criteria document
$CD4^+$	helper T lymphocyte
$CD8^+$	suppressor T lymphocyte
CDC	Centers for Disease Control and Prevention
CEPEX	Central Equatorial Pacific Experiment
CFD	Computational Fluid Dynamics
CG	cloud-to-ground (flash)
cGMP	cyclic guanosine-3',5'-monophosphate
CH_4	methane
C_2H_4	ethene
C_2H_6	ethane
C_5H_8	isoprene
CHAD	Consolidated Human Activities Database
CH_3-CHO	acetaldehyde
$CH_3CH(O)OONO_2$	peroxyacetyl nitrate
CH_3CN	acetonitrile
CH_3-C(O)	acetyl radical
CH_3-C(O)H	acetaldehyde
$CH_3C(O)O$	peroxyacetyl radical
CH_3-$C(O)O_2$, CH_3−C(O)OO	acetyl peroxy, peroxyacetyl
CH_3OOH	methyl hydroperoxide
$(CH_3)_2S$	dimethylsulfide
CH_3-S-H	methyl mercaptan
CH_3SO_3H	methanesulfonic acid
CH_3-S-S-CH_3	dimethyl disulfide
Cl	chorine
Cl^-	chorine ion
CLaMS	type of Lagrangian model
$ClNO_2$	nitryl chloride
$ClNO_3$	chlorine nitrate
CMAQ	Community Model for Air Quality;
CMAQ	Community Multiscale Air Quality (model)
CMSA	consolidated metropolitan statistical area
CO	carbon monoxide
COD	coefficient of divergence

COPD	chronic obstructive pulmonary disease
CPBM	Canyon Plume-Box Model
CS_2	carbon disulfide
CTM	chemistry transport model
DEP	diesel exhaust particulates
DEPcCBP	diesel exhaust particulates extract-coated carbon black particles
DL	detection limit
DMN	dimethylnitrosamine
DMSO	dimethylsulfoxide
DNA	deoxyribonucleic acid
DNS	Direct Numerical Simulation
DOAS	differential optical absorption spectroscopy
DU	Dobson units
EC	molecular carbon
ECRHS	European Community Respiratory Health Survey
EDMAS	Exposure and Dose Modeling and Analysis System
EDXRF	energy dispersive X-ray fluorescence
EE	energy expenditure
eGPx	extracellular glutathione peroxidase
ELF	epithelial lining fluid
EMD	Ecole des Mines de Douai (laboratory)
EPA	U.S. Environmental Protection Agency
ER	emergency room
ESR	electron spin resonance (spectroscopy)
ETS	environmental tobacco smoke
EXPOLIS	Air Pollution Exposure Distributions of Adult Urban Populations in Europe
FEV_1	forced expiratory volume in 1 second
FL	fluoranthene
FLEXPART	type of Lagrangian model
FPD	flame photometric detection
FT	free troposphere
FTIR	Fourier Transform Infrared Spectroscopy
FVC	forced vital capacity
FW2	black carbon soot model
γGCS	gamma-glutamylcysteine synthetase
γGT	glutamyltranspeptidase
γN_2O_5	uptake coefficient for N_2O_5

GC/ECD	gas chromatography-electron capture detection
GEE	Generalized Estimating Equations
GEOS-CHEM	three-dimensional model of atmospheric composition driven by assimilated Goddard Earth Orbiting System observations
GEOS-1 DAS	NASA Goddard Earth Orbiting System Data Assimilation System
GIS	Geographic Information System
GMP	guanosine-3',5'-monophosphate
GOME	Global Ozone Monitoring Instrument
GS	glutathione synthetase
GSH	glutathione; reduced glutathione
GSSG	oxidized glutathione; glutathione disulfide
GST	glutathione *S*-transferase (e.g., GST M1, GST P1, GST T1)
H^+	hydrogen ion
H_2	molecular hydrogen; hydrogen gas
HAPEM	Hazardous Air Pollutant Exposure Model
HCHO	formaldehyde
HCl	hydrochloric acid
HCN	hydrogen cyanide
HCs	hydrocarbons
HEADS	Harvard-EPA Annular Denuder System
5-HETE	5-hydroxyeicosatetraenoic acid
HNO_3	nitric acid
HNO_4	pernitric acid
HO_2	hydroperoxyl; hydroperoxy radical
H_2O_2	hydrogen peroxide
HOBr	hypobromous acid
HOCl	hypochlorous acid
HONO, HNO_2	nitrous acid
HO_2NO_2	peroxynitric acid
HOX	hypohalous acid
H_2S	hydrogen sulfide
HSO_3	hydrogen sulfite ion
HSO_3^-	hydrogen sulfite
HSO_4^-	bisulfate ion
H_2SO_4	sulfuric acid
hν	solar ultraviolet photon
I	iodine
I_2	molecular iodine

IBEM	Individual Based Exposure Models
IC	intracloud (flash); ion chromatography
ICAM-1	intercellular adhesion molecule-1
ICARTT	International Consortium for Atmospheric Research on Transport and Transformation
Ig	immunoglobulin (e.g., IgA, IgE, IgG)
IIASA	International Institute for Applied Systems Analysis
IL	interleukin (e.g., IL-1, IL-6, IL-8)
IMPROVE	Interagency Monitoring of Protected Visual Environments
INDAIR	(model)
INO_3	iodine nitrate
INTEX-NA	NASA Intercontinental Chemical Transport Experiment - North America
IQR	interquartile range
JPL	Jet Propulsion Laboratory
K_a	acid dissociation constant in M
K_H	Henry's Law constant in M atm^{-1}
KH	potassium hydride
K_w	ion product of water
LDH	lactic acid dehydrogenase
LES	Large Eddy Simulation
LIF	laser-induced fluorescence
LP	long-path
LPG	liquified propane gas
LT	leukotriene (e.g., LTB_4, LTC_4, LTD_4, LTE_4)
LWC	liquid water content
M	air molecule
MAQSIP	Multiscale Air Quality Simulation Platform
MAX	multi axis
MBL	marine boundary layer
MCM	master chemical mechanism
MEM	model ensemble mean
MENTOR-1A	Modeling Environment for Total Risk for One-Atmosphere studies
MET	metabolic equivalent of work
MgO	magnesium oxide
MIESR	matrix isolation electron spin resonance (spectroscopy)
MM5	National Center for Atmospheric Research/Penn State Mesoscale Model
MOBILE6	Highway Vehicle Emission Factor Model

MoO_x	molybdenum oxide
MOZART-2	(model)
MPAN	peroxymethacryloyl nitrate; peroxy-methacrylic nitric anhydride
mRNA	messenger ribonucleic acid
MSA	metropolitan statistical area
^{15}N	nitrogen-15 radionuclide
N	nitrogen
N, n	number of observations
NA, N/A, N.A.	not available
NAAQS	National Ambient Air Quality Standards
Na_2CO_3	sodium carbonate
NADP	National Atmospheric Deposition Program
NADPH	reduced nicotinamide adenine dinucleotide phosphate
$NaHCO_3$	sodium bicarbonate
NARSTO	North American Regional Strategy for Atmospheric Ozone
NASA	National Aeronautics and Space Administration
NCAR	National Center for Atmospheric Research
NDMA	*N*-nitrosodimethylamine
NEM	National Ambient Air Quality Standards Exposure Model
NERL	National Exposure Research Laboratory
2NF	2-nitrofluoranthene
NH_2	amino
NH_3	ammonia
NH_4^+	ammonium ion
NH_4Cl	ammonium chloride
NHLBI	U.S. National Heart, Lung and Blood Institute
NH_4NO_3	ammonium nitrate?
$(NH_4)_2SO_4$	ammonium sulfate
NIST	National Institute of Standards and Technology
NK	natural killer (lymphocytes)
NMHCs	nonmethane hydrocarbons
NMOCs	nonmethane organic compounds
NMOR	*N*-nitrosomorpholine
NO	nitric oxide
NO_2	nitrogen dioxide
NO_2^-	nitrite
NO_3	nitrate (radical)
NO_3^-	nitrate

N_2O_5	dinitrogen pentoxide
NO_x	nitrogen oxides; oxides of nitrogen
NO_y	sum of NO_x and NO_z; odd nitrogen species
NO_z	oxides of nitrogen and nitrates; difference between NO_y and NO_x
NP	national park
1NP	1-nitropyrene
NPAHs	nitro polycyclic aromatic hydrocarbons
NR	data not relevant
NR, N.R., N/R	not reported
NRC	National Research Council
NSA	nitrosating agent
nss	non-sea-salt
NTRMs	NIST Traceable Reference Materials
^{16}O	oxygen-16 radionuclide
O_3	ozone
OAQPS	Office of Air Quality Planning and Standards
OC	organic carbon
OCS	carbonyl sulfide
$O(^1D)$	electronically excited oxygen atom
OH	hydroxyl radical
OMI	Ozone Monitoring Instrument
$O(^3P)$	ground-state oxygen atom
OPE	ozone production efficiency
OSPM	Danish Operational Street Pollution Model
O_x	odd oxygen species; total oxidants
$P(HNO_3)$	particulate nitrate
P, p	probability value
P_{90}	values of the 90th percentile absolute difference in concentrations
PAHs	polycyclic aromatic hydrocarbons
PAMS, PAMs	Photochemical Aerometric Monitoring System
PAN	peroxyacetyl nitrate; peroxyacyl nitrate
P_aO_2	partial pressure of arterial oxygen
PAQSMs	photochemical air quality simulation models
PAR	proximal alveolar region
PBEM	Population Based Exposure Models
PIXE	particle induced X-ray emission
PM	particulate matter
PM_{10}	combination of coarse and fine particulate matter

$PM_{10-2.5}$	coarse particulate matter
$PM_{2.5}$	fine particulate matter
PMA	phorbol myristate acetate
PM-CAMx	Particulate Matter Comprehensive Air Quality Model with Extensions
PMN	polymorphonuclear leukocytes
PMT	photomultiplier tube
pNEM	Probabilistic National Ambient Air Quality Standard Exposure Model
POM	particulate organic matter
ppb	parts per billion
ppbv	parts per billion by volume
ppm × h	parts per million × hours
ppm	parts per million
PPN	peroxypropionyl nitrate; peroxypropionic nitric anhydride
ppt	parts per trillion
pptv	parts per trillion by volume
PRB	policy relevant background
psi	pounds per square inch
PTEAM	Particle Total Exposure Assessment Methodology (study)
PTEP	PM_{10} Technical Enhancement Program
PTFE	polytetrafluoroethylene (Teflon)
PY	pyrene
r	correlation coefficient
R^2	coefficient of determination
RACM	Regional Air Chemistry Mechanism
RADM	Regional Acid Deposition Model
RAMs	Regional Atmospheric Modeling System
RANS	Reynolds Averaged Numerical Simulation
RAPS	Regional Air Pollution Study
RBC	red blood cell
RCS	Random Component Superposition (model)
RDBMS	Relational Database Management Systems
REHEX	Regional Human Exposure Model
RH	relative humidity
RIOPA	Relationship of Indoor, Outdoor, and Personal Air (study)
RMR	resting metabolic rate
RNO_2	nitro compounds
RO_2	organic peroxyl; organic peroxy

$RONO_2$	organic nitrate
$ROONO_2$, RO_2NO_2	peroxy nitrate
ROS	reactive oxygen species
r_p	Pearson correlation coefficient
r_s	Spearman rank correlation coefficient
RSD	relative standard deviation
σ	sigma; standard deviation
^{34}S	sulfur-34 radionuclide
S_2*	electronically excited sulfur molecules
S_2O	disulfur monoxide
SAPALDIA	Study of Air Pollution and Lung Diseases in Adults
SCE	sister chromatid exchange
SCIAMACHY	Scanning Imaging Absorption Spectrometer for Atmospheric Chartography
SCOS97	1997 Southern California Ozone Study
SGV	subgrid variability
SHEDS	Simulation of Human Exposure and Dose System
SMOKE	Spare-Matrix Operator Kernel Emissions (system)
SO	sulfur monoxide
SO_2	sulfur dioxide
SO_3	sulfur trioxide
SOA	secondary organic aerosol
SONEX	Subsonic Assessment Ozone and Nitrogen Oxides Experiment
SOS	Southern Oxidant Study
SP	surfactant protein (e.g., SP-A, SP-D)
SRM	standard reference material
STE	stratospheric-tropospheric exchange
STEP	Stratospheric-Tropospheric-Exchange Project
STN	Speciation Trends Network
STPD	standard temperature and pressure, dry
STREET	type of street canyon model
STRF	Spatio-Temporal Random Field (theory)
τ	tau; atmospheric lifetime
T	time; duration of exposure
TAR	Third Assessment Report
TBA	thiobarbituric acid
TDLAS	tunable-diode laser absorption spectroscopy
TEA	triethanolamine

TexAQS	Texas Air Quality Study
Tg	teragram
THEES	Total Human Environmental Exposure Study
TNF	tumor necrosis factor (e.g., TNF-α)
TOR	thermal-optical reflectance
Torr	unit of pressure
TRACE-P	Transport and Chemical Evolution over the Pacific
TTFMS	two-tone frequency-modulated spectroscopy
TVOCs	total volatile organic compounds
TX	thromboxane (e.g., TXA_2, TXB_2)
UAM	Urban Airshed Model
UMD-CTM	University of Maryland Chemical Transport Model
UV	ultraviolet
V_E	total ventilation rate
VESTA	Five (V) Epidemiological Studies on Transport and Asthma
VOC	volatile organic compound
V_T	tidal volume
WHO	World Health Organization
XRF	X-ray fluorescence

AX1. CHAPTER 1 ANNEX – INTRODUCTION

The draft Annexes are prepared in support of the draft Integrated Science Assessment for Oxides of Nitrogen – Health Criteria (EPA/600/R-07/093). The Integrated Science Assessment (ISA) presents a concise synthesis of the most policy-relevant science to form the scientific foundation for the review of the primary (health-based) national ambient air quality standards (NAAQS) for nitrogen dioxide (NO_2). This series of Annexes provide more extensive and detailed summaries of the most pertinent scientific literature. The Annexes identify, evaluate, and summarize scientific research in the areas of atmospheric sciences, air quality analyses, exposure assessment, dosimetry, controlled human exposure studies, toxicology, and epidemiology, focusing on studies relevant to the review of the primary NAAQS.

These draft Annexes are organized by scientific study areas and include research that is relevant to the key policy questions discussed previously to provide an evidence base supporting the development of the ISA, risk, and exposure assessments. In Annex 1, we provide legislative background and history of previous reviews of the NAAQS for oxides of nitrogen. In Annex 2, we present evidence related to the physical and chemical processes controlling the production, destruction, and levels of reactive nitrogen compounds in the atmosphere, including both oxidized and reduced species. Annex 3 presents information on environmental concentrations, patterns, and human exposure to ambient oxides of nitrogen; however, most information relates to NO_2. Annex 4 presents results from toxicological studies as well as information on dosimetry of oxides of nitrogen. Annex 5 discusses results from controlled human exposure studies, and Annex 6 discusses evidence from epidemiological studies. These Annexes include more detailed information on health or exposure studies that is summarized in tabular form, as well as more extensive discussion of atmospheric chemistry, source, exposure, and dosimetry information. Annex tables for health studies are generally organized to include information about (1) concentrations of oxides of nitrogen levels or doses and exposure times, (2) description of study methods employed, (3) results and comments, and (4) quantitative outcomes for oxides of nitrogen measures.

AX1.1 LEGISLATIVE REQUIREMENTS

Two sections of the Clean Air Act (CAA) govern the establishment and revision of the national ambient air quality standards (NAAQS). Section 108 (U.S. Code, 2003a) directs the Administrator to identify and list "air pollutants" that "in his judgment, may reasonably be anticipated to endanger public health and welfare" and whose "presence in the ambient air results from numerous or diverse mobile or stationary sources" and to issue air quality criteria for those that are listed. Air quality criteria are intended to "accurately reflect the latest scientific knowledge useful in indicating the kind and extent of identifiable effects on public health or welfare which may be expected from the presence of [a] pollutant in ambient air."

Section 109 (U.S. Code, 2003b) directs the Administrator to propose and promulgate "primary" and "secondary" NAAQS for pollutants listed under Section 108. Section 109(b) (1) defines a primary standard as one "the attainment and maintenance of which in the judgment of the Administrator, based on such criteria and allowing an adequate margin of safety, are requisite to protect the public health."[1] A secondary standard, as defined in Section 109(b)(2), must "specify a level of air quality the attainment and maintenance of which, in the judgment of the Administrator, based on such criteria, is required to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air."[2]

The requirement that primary standards include an adequate margin of safety was intended to address uncertainties associated with inconclusive scientific and technical information available at the time of standard setting. It was also intended to provide a reasonable degree of protection against hazards that research has not yet identified. See Lead Industries Association v. EPA, 647 F.2d 1130, 1154 (D.C. Cir 1980), cert. denied, 449 U.S. 1042 (1980); American Petroleum Institute v. Costle, 665 F.2d 1176, 1186 (D.C. Cir. 1981), cert. denied, 455 U.S. 1034 (1982). Both kinds of uncertainties are components of the risk associated with pollution at levels below those at which human health effects can be said to occur with

[1] The legislative history of Section 109 indicates that a primary standard is to be set at "the maximum permissible ambient air level … which will protect the health of any [sensitive] group of the population" and that, for this purpose, "reference should be made to a representative sample of persons comprising the sensitive group rather than to a single person in such a group" [U.S. Senate (1970)].

[2] Welfare effects as defined in Section 302(h) [U.S. Code, 2005)] include, but are not limited to, "effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being."

reasonable scientific certainty. Thus, in selecting primary standards that include an adequate margin of safety, the Administrator is seeking not only to prevent pollution levels that have been demonstrated to be harmful but also to prevent lower pollutant levels that may pose an unacceptable risk of harm, even if the risk is not precisely identified as to nature or degree.

In selecting a margin of safety, the U.S. Environmental Protection Agency (EPA) considers such factors as the nature and severity of the health effects involved, the size of sensitive population(s) at risk, and the kind and degree of the uncertainties that must be addressed. The selection of any particular approach to providing an adequate margin of safety is a policy choice left specifically to the Administrator's judgment. See Lead Industries Association v. EPA, supra, 647 F.2d at 1161-62.

In setting standards that are "requisite" to protect public health and welfare, as provided in Section 109(b), EPA's task is to establish standards that are neither more nor less stringent than necessary for these purposes. In so doing, EPA may not consider the costs of implementing the standards. See generally Whitman v. American Trucking Associations, 531 U.S. 457, 465-472, and 475-76 (U.S. Supreme Court, 2001).

Section 109(d)(1) requires that "not later than December 31, 1980, and at 5-year intervals thereafter, the Administrator shall complete a thorough review of the criteria published under Section 108 and the national ambient air quality standards and shall make such revisions in such criteria and standards and promulgate such new standards as may be appropriate …." Section 109(d)(2) requires that an independent scientific review committee "shall complete a review of the criteria … and the national primary and secondary ambient air quality standards … and shall recommend to the Administrator any new standards and revisions of existing criteria and standards as may be appropriate …." Since the early 1980s, this independent review function has been performed by the Clean Air Scientific Advisory Committee (CASAC) of EPA's Science Advisory Board.

AX1.2 HISTORY OF REVIEWS OF THE PRIMARY NAAQS FOR NO_2

On April 30, 1971, EPA promulgated identical primary and secondary NAAQS for nitrogen dioxide (NO_2), under Section 109 of the Act, set at 0.053 parts per million (ppm), annual average (Federal Register, 1971). In 1982, EPA published Air Quality Criteria for Oxides of Nitrogen (1982 NO_x AQCD) (U.S. Environmental Protection Agency, 1982), which

updated the scientific criteria upon which the initial NO_2 standards were based. On February 23, 1984, EPA proposed to retain these standards (Federal Register, 1984). After taking into account public comments, EPA published the final decision to retain these standards on June 19, 1985 (Federal Register, 1985).

On July 22, 1987, EPA announced that it was undertaking plans to revise the 1982 NO_x air quality criteria (Federal Register, 1987). In November 1991, EPA released an updated draft air quality criteria document (AQCD) for CASAC and public review and comment (Federal Register, 1991). The draft document provided a comprehensive assessment of the available scientific and technical information on heath and welfare effects associated with NO_2 and other oxides of nitrogen. The CASAC reviewed the document at a meeting held on July 1, 1993, and concluded in a closure letter to the Administrator that the document "provides a scientifically balanced and defensible summary of current knowledge of the effects of this pollutant and provides an adequate basis for EPA to make a decision as to the appropriate NAAQS for NO_2" (Wolff, 1993).

The EPA also prepared a draft Staff Paper that summarized and integrated the key studies and scientific evidence contained in the revised AQCD and identified the critical elements to be considered in the review of the NO_2 NAAQS. The Staff Paper received external review at a December 12, 1994 CASAC meeting. CASAC comments and recommendations were reviewed by EPA staff and incorporated into the final draft of the Staff Paper as appropriate. CASAC reviewed the final draft of the Staff Paper in June 1995 and responded by written closure letter (Wolff, 1995). In September of 1995, EPA finalized the document entitled, "Review of the National Ambient Air Quality Standards for Nitrogen Dioxide Assessment of Scientific and Technical Information" (U.S. Environmental Protection Agency, 1995).

Based on that review, the Administrator announced her proposed decision not to revise either the primary or the secondary NAAQS for NO_2 (Federal Register, 1995). The decision not to revise the NO_2 NAAQS was finalized after careful evaluation of the comments received on the proposal. The level for both the existing primary and secondary NAAQS for NO_2 is 0.053 ppm annual arithmetic average, calculated as the arithmetic mean of the 1-h NO_2 concentrations.

AX1.3 REFERENCES

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AX2. CHAPTER 2 ANNEX – ATMOSPHERIC CHEMISTRY OF NITROGEN AND SULFUR OXIDES

AX2.1 INTRODUCTION

Nitrogen oxides (NO_x) along with volatile organic compounds (VOCs) including anthropogenic and biogenic hydrocarbons, aldehydes, etc. and carbon monoxide (CO) serve as precursors in the formation of ozone (O_3) and other elements of photochemical smog. Nitrogen oxides are defined here as nitric oxide (NO) and nitrogen dioxide (NO_2), the latter of which is a U.S. EPA Criteria Air Pollutant; similarly, oxides of sulfur (SO_x) are defined here to be sulfur monoxide (SO), sulfur dioxide (SO_2), the largest component of SO_x and also a U.S. EPA Criteria Air Pollutant, and sulfur trioxide (SO_3). SO_3 rapidly reacts with water vapor to form H_2SO_4, and only SO_2 is present in the atmosphere at detectable levels.

Nitrogen dioxide is an oxidant and can further react to form other photochemical oxidants, in particular the organic nitrates, including peroxy acetyl nitrates (PAN) and higher PAN analogues. It can also react with toxic compounds such as polycyclic aromatic hydrocarbons (PAHs) to form nitro-PAHs, which may be even more toxic than the precursors. Nitrogen dioxide together with sulfur dioxide (SO_2), another U.S. EPA criteria air pollutant, can be oxidized to the strong mineral acids, nitric acid (HNO_3) and sulfuric acid (H_2SO_4), which contribute to the acidity of cloud, fog, and rainwater, and can form ambient particles.

The role of NO_x in O_3 formation was reviewed in Chapter 2 (Section 2.2) of the latest AQCD for Ozone and Other Photochemical Oxidants (U.S. Environmental Protection Agency, 2006 CD06), and in numerous texts (e.g., Seinfeld and Pandis, 1998; Jacob, 2000; Jacobson, 2002). Mechanisms for transporting O_3 precursors, the factors controlling the efficiency of O_3 production from NO_x, methods for calculating O_3 from its precursors, and methods for measuring NO_x were all reviewed in Section 2.6 of CD06. The main points from those discussions in CD06 and updates, based on new materials will be presented here. Ammonia (NH_3) is included here because its oxidation can be a source of NO_x, and it is a precursor for ammonium ions (NH_4^+), which play a key role in neutralizing acidity in ambient particles and in cloud, fog, and rain water. Ammonia is also involved in the ternary nucleation of new particles, and it reacts with gaseous HNO_3 to form ammonium nitrate (NH_4NO_3), which is a major

constituent of ambient Particulate Matter (PM) in many areas. Ammonia is also involved in over nitrification of aqueous and terrestrial ecosystems and participates in the N cascade (Galloway et al., 2003)

The atmospheric chemistry of NO_x is discussed in Section AX2.2, and of SO_2 in Section AX2.3. Mechanisms for the formation of aqueous-phase sulfate (SO_4^{2-}) and nitrate (NO_3^-) are reviewed in Section AX2.4. Sources and emissions of NO_x, NH_3, and SO_2 are discussed in Section AX2.5. Modeling methods used to calculate the atmospheric chemistry, transport, and fate of NO_x and SO_2 and their oxidation products are presented in Section AX2.6. Measurement techniques for the nitrogen-containing compounds and for SO_2, nitrates, sulfates, and ammonium ion are discussed in Section AX2.8. Estimates of policy-relevant background concentrations of NO_x and SO_x are given in Section AX2.9. An overall review of key points in this chapter is given in Section AX2.11.

The overall chemistry of reactive nitrogen compounds in the atmosphere is summarized in Figure AX2-1 and is described in greater detail in the following sections. Nitrogen oxides are emitted primarily as NO with smaller quantities of NO_2. Emissions of NO_x are spatially distributed vertically with some occurring at or near ground level and others aloft as indicated in Figure AX2-1. Because of atmospheric chemical reactions, the relative abundance of different compounds contributed by different sources varies with location. Both anthropogenic and natural (biogenic) processes emit NO_x. In addition to gas phase reactions, multiphase processes are important for forming aerosol-phase pollutants, including aerosol NO_3^-.

AX2.2 CHEMISTRY OF NITROGEN OXIDES IN THE TROPOSPHERE

AX2.2.1 Basic Chemistry

There is a rapid photochemical cycle in the troposphere that involves photolysis of NO_2 by solar UV-A radiation to yield NO and a ground-state oxygen atom, $O(^3P)$:

$$NO_2 + h\nu \rightarrow NO + O(^3P), \qquad \text{(AX2-1)}$$

This ground-state oxygen atom can then combine with molecular oxygen (O_2) to form O_3; and, colliding with any molecule from the surrounding air (M = N_2, O_2, etc), the newly formed O_3 molecule, transfers excess energy and is stabilized:

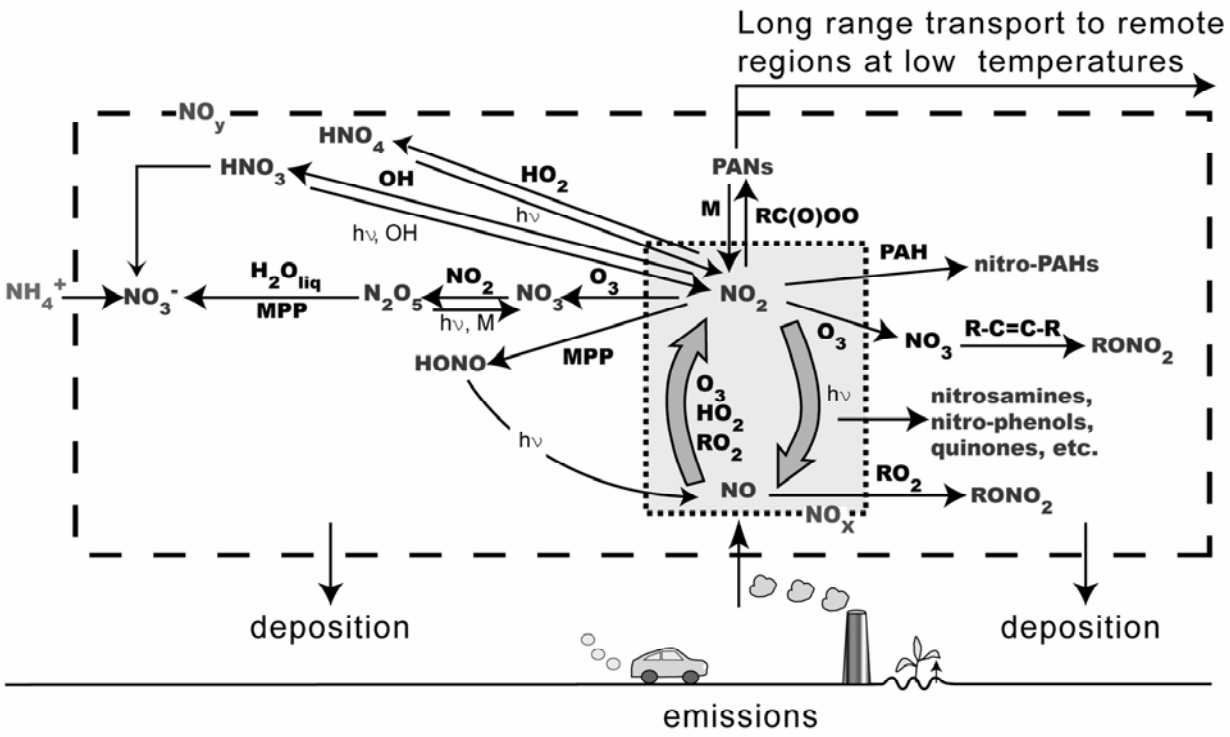


Figure AX2-1. Schematic diagram of the cycle of reactive nitrogen species in the atmosphere. MPP refers to multi-phase process; hν to a photon of solar energy.

$$O(^3P) + O_2 + M \rightarrow O_3 + M, \qquad \text{(AX2-2)}$$

where M = N_2, O_2. Reaction AX2-2 is the only significant reaction forming O_3 in the troposphere.

NO and O_3 react to reform NO_2:

$$NO + O_3 \rightarrow NO_2 + O_2. \qquad \text{(AX2-3)}$$

Reaction AX2-3 is responsible for O_3 decreases and NO_2 increases found near sources of NO (e.g., highways), especially at night when the actinic flux is 0. Oxidation of reactive VOCs leads to the formation of reactive radical species that allow the conversion of NO to NO_2 without the participation of O_3 (as in Reaction AX2-3):

$$NO \xrightarrow{HO_2,\ RO_2} NO_2. \quad \text{(AX2-4)}$$

Ozone, therefore, can accumulate as NO_2 photolyzes as in Reaction AX2-1, followed by Reaction AX2-2. Specific mechanisms for the oxidation of a number of VOCs were discussed in the O_3 AQCD (U.S. Environmental Protection Agency, 2006).

It is often convenient to speak about families of chemical species defined in terms of members that interconvert rapidly among themselves on time scales that are shorter than those for formation or destruction of the family as a whole. For example, an "odd oxygen" (O_x) family can be defined as

$$O_x = \sum(O(^3P) + O(^1D) + O_3 + NO_2)$$

In much the same way, NO_x is sometimes referred to as "odd nitrogen". Hence, we see that production of O_x occurs by the schematic Reaction AX2-4, and that the sequence of reactions given by reactions AX2-1 through AX2-3 represents no net production of O_x. Definitions of species families and methods for constructing families are discussed in Jacobson (1999) and references therein. Other families that include nitrogen-containing species (and which will be referred to later in this chapter) include:

$$NO_x = (NO + NO_2),$$

One can then see that production of O_x occurs by the schematic Reaction AX2-4, and that the sequence of reactions given by reactions AX2-1 through AX2-3 represents no net production of O_x. Definitions of species families and methods for constructing families are discussed in Jacobson (1999) and references therein. Other families that include nitrogen-containing species, and which will be referred to later in this chapter, are: (which is the sum of the products of the oxidation of NO_x)

$$NO_Z = \sum (HNO_3 + HNO_4 + NO_3 + 2NO_2O_5 + PAN(CH_3CHO\text{ - }OO\text{ - }NO_2) + \textit{other organic nitraties} + \textit{halogen nitrates} + \textit{particulate nitrate});$$

$$NO_y = NO_x + NO_z + HONO;$$
$$\textit{and } NH_x = NH_3 + NH_4^+$$

The reaction of NO_2 with O_3 leads to the formation of NO_3^- radical,

$$NO_2 + O_3 \rightarrow NO_3 + O_2, \quad \text{(AX2-5)}$$

However, because the NO_3 radical photolyzes rapidly (lifetime of ≈5 s during the photochemically most active period of the day around local solar noon (Atkinson et al., 1992a),

$$NO_3 + h\nu \rightarrow NO + O_{2(10\%)} \quad \text{(AX2-6a)}$$

$$NO_2 + O(^3P)_{(90\%)} \quad \text{(AX2-6b)}$$

its concentration remains low during daylight hours, but can increase after sunset to nighttime concentrations of $<5 \times 10^7$ to 1×10^{10} molecules cm^{-3} (<2 to 430 parts per trillion (ppt)) over continental areas influenced by anthropogenic emissions of NO_x (Atkinson et al., 1986). At night, NO_3, rather than the hydroxyl radical (OH), is the primary oxidant in the system.

Nitrate radicals can combine with NO_2 to form dinitrogen pentoxide (N_2O_5):

$$NO_3 + NO_2 \overset{M}{\longleftrightarrow} N_2O_5 \quad \text{(AX2-7)}$$

and N_2O_5 both photolyzes and thermally decomposes back to NO_2 and NO_3 during the day; however, N_2O_5 concentrations ($[N_2O_5]$) can accumulate during the night to parts per billion (ppb) levels in polluted urban atmospheres.

The tropospheric chemical removal processes for NO_x include reaction of NO_2 with the OH radical and hydrolysis of N_2O_5 in aqueous aerosol solutions if there is no organic coating. Both of these reactions produce HNO_3.

$$OH + NO_2 \xrightarrow{M} HNO_3 \quad \text{(AX2-8)}$$

$$N_2O_5 \xrightarrow{H_20(1)} HNO_3 \quad \text{(AX2-9)}$$

The gas-phase reaction of the OH radical with NO_2 (Reaction AX2-8) initiates one of the major and ultimate removal processes for NO_x in the troposphere. This reaction removes OH and NO_2 radicals and competes with hydrocarbons for OH radicals in areas characterized by high NO_x concentrations, such as urban centers (see Section AX2.2.2). The timescale (τ) for

conversion of NO_x to HNO_3 in the planetary boundary layer at 40 N latitude ranges from about 4 hours in July to about 16 hours in January. The corresponding range in τ at 25 N latitude is between 4 and 5 hours, while at 50 N latitude, HNO_3 τ ranges from about 4 to 20 hours (Martin et al., 2003). In addition to gas-phase HNO_3, Golden and Smith (2000) have shown on the basis of theoretical studies that pernitrous acid (HOONO) is also produced by the reaction of NO_2 and OH radicals. However, this channel of production most likely represents a minor yield (approximately 15% at the surface) (Jet Propulsion Laboratory, 2003). Pernitrous acid will also thermally decompose and can photolyze. Gas-phase HNO_3 formed from Reaction AX2-8 undergoes wet and dry deposition to the surface, and uptake by ambient aerosol particles. Reaction AX2-8 limits NO_x τ to a range of hours to days.

In addition to the uptake of HNO_3 on particles and in cloud drops, it photolyzes and reacts with OH radicals via

$$HNO_3 + h\nu \rightarrow OH + NO_2 \qquad \text{(AX2-10)}$$

and

$$HNO_3 + OH \rightarrow NO_3 + H_2O. \qquad \text{(AX2-11)}$$

The lifetime of HNO_3 with respect to these two reactions is long enough for HNO_3 to act as a reservoir species for NO_x during long-range transport, contributing in this way to NO_2 levels and to O_3 formation in areas remote from the source region of the NO_x that formed this HNO_3.

Geyer and Platt (2002) concluded that Reaction AX2-9 constituted about 10% of the removal of NO_x at a site near Berlin, Germany during spring and summer. However, other studies found a larger contribution to HNO_3 production from Reaction AX2-9. Dentener and Crutzen (1993) estimated 20% in summer and 80% of HNO_3 production in winter is from Reaction AX2-9. Tonnesen and Dennis (2000) found between 16 to 31% of summer HNO_3 production was from Reaction AX2-9. The contribution of Reaction AX2-9 to HNO_3 formation is highly uncertain during both winter and summer. The importance of Reaction AX2-9 could be much higher during winter than during summer because of the much lower concentration of OH radicals and the enhanced stability of N_2O_5 due to lower temperatures and less sunlight. Note that Reaction AX2-9 proceeds as a heterogeneous reaction. Recent work in the northeastern U.S.

indicates that this reaction is proceeds at a faster rate in power plant plumes than in urban plumes (Brown et al., 2006a,b; Frost et al., 2006).

OH radicals also can react with NO to produce nitrous acid (HONO or HNO_2):

$$OH + NO \xrightarrow{M} HNO_2. \quad \text{(AX2-12)}$$

In the daytime, HNO_2 is rapidly photolyzed back to the original reactants:

$$HNO_2 + h\nu \rightarrow OH + NO. \quad \text{(AX2-13)}$$

Reaction AX2-12 is, however, a negligible source of HONO, which is formed mainly by multiphase processes (see Section AX2.2.3). At night, heterogeneous reactions of NO_2 in aerosols or at the earth's surface result in accumulation of HONO (Lammel and Cape, 1996; Jacob, 2000; Sakamaki et al., 1983; Pitts et al., 1984; Svensson et al., 1987; Jenkin et al., 1988; Lammel and Perner, 1988; Notholt et al., 1992a,b). Harris et al. (1982) and Zhang et al. (2006) (e.g.) suggested that photolysis of this HNO_2 at sunrise could provide an important early-morning source of OH radicals to drive O_3 formation

Hydroperoxy (HO_2) radicals can react with NO_2 to produce pernitric acid (HNO_4):

$$HO_2 + NO_2 + M \rightarrow HNO_4 + M \quad \text{(AX2-14)}$$

which then can thermally decompose and photolyze back to its original reactants. The acids formed in these gas-phase reactions are all water soluble. Hence, they can be incorporated into cloud drops and in the aqueous phase of particles.

Although the lifetimes of HNO_4 and N_2O_5 are short (minutes to hours) during typical summer conditions, they can be much longer at the lower temperatures and darkness found during the polar night. Under these conditions, species such as PAN, HNO_3, HNO_4, and N_2O_5 serve as NO_x reservoirs that can liberate NO_2 upon the return of sunlight during the polar spring. A broad range of organic nitrogen compounds can be directly emitted by combustion sources or formed in the atmosphere from NO_x emissions. Organic nitrogen compounds include the PANs, nitrosamines, nitro-PAHs, and the more recently identified nitrated organics in the quinone family. Oxidation of VOCs produces organic peroxy radicals (RO_2), as discussed in the latest AQCD for Ozone and Other Photochemical Oxidants (U.S. Environmental Protection Agency,

2006). Reaction of RO_2 radicals with NO and NO_2 produces organic nitrates ($RONO_2$) and peroxynitrates (RO_2NO_2):

$$RO_2 + NO \xrightarrow{M} RONO_2 \quad \text{(AX2-15)}$$

$$RO_2 + NO_2 \xrightarrow{M} RO_2NO_2 \quad \text{(AX2-16)}$$

Reaction (AX2-15) is a minor branch for the reaction of RO_2 with NO. The major branch produces RO and NO_2, as discussed in the next section; however, the organic nitrate yield increases with carbon number (Atkinson, 2000).

The most important of these organic nitrates is PAN, the dominant member of the broader family of peroxyacylnitrates which includes peroxypropionyl nitrate (PPN) of anthropogenic origin and peroxymethacrylic nitrate (MPAN) produced from isoprene oxidation. The PANs are formed by the combination reaction of acetyl peroxy radicals with NO_2:

$$CH_3C(O)\text{–}OO + NO_2 \rightarrow CH_3C(O)OONO_2 \quad \text{(AX2-17)}$$

where the acetyl peroxy radicals are formed mainly during the oxidation of ethane (C_2H_6). Acetaldehyde (CH_3CHO) is formed as an intermediate species during the oxidation of ethane. Acetaldehyde can be photolyzed or react with OH radicals to yield acetyl radicals:

$$CH_3\text{–}C(O)H + h\nu \rightarrow CH_3\text{–}C(O) + H \quad \text{(AX2-18)}$$

$$CH_3\text{–}C(O)H + OH \rightarrow CH_3\text{–}C(O) + H_2O \quad \text{(AX2-19)}$$

Acetyl radicals then react with O_2 to yield acetyl peroxy radicals

$$CH_3\text{–}C(O) + O_2 + M \rightarrow CH_3C(O)\text{–}OO + M \quad \text{(AX2-20)}$$

However, acetyl peroxy radicals will react with NO in areas of high NO concentrations

$$CH_3(CO)\text{–}OO + NO \rightarrow CH_3(CO)\text{–}O + NO_2 \quad \text{(AX2-21)}$$

and the acetyl-oxy radicals will then decompose

$$CH_3(CO)\text{–}O \rightarrow CH_3 + CO_2 \quad \text{(AX2-22)}$$

Thus, the formation of PAN is favored at conditions of high ratios of NO_2 to NO, which are most typically found under low NO_x conditions. The PANs both thermally decompose and photolyze back to their reactants on timescales of a few hours during warm sunlit conditions, with lifetimes with respect to thermal decomposition ranging from ~1 hour at 298 K to ~2.5 days at 273 K, up to several weeks at 250 K. Thus, they can provide an effective sink of NO_x at cold temperatures and high solar zenith angles, allowing release of NO_2 as air masses warm, in particular by subsidence. The PANs are also removed by uptake to vegetation (Sparks et al., 2003; Teklemariam and Sparks, 2004).

The organic nitrates may react further, depending on the functionality of the R group, but they will typically not return NO_x and can therefore be viewed mainly as a permanent sink for NO_x, as alkyl nitrates are sparingly soluble and will photolyze. This sink is usually small compared to HNO_3 formation, but the formation of isoprene nitrates may be a significant sink for NO_x in the United States in summer (Liang et al., 1998).

The peroxynitrates produced by (1-16) are thermally unstable and most have very short lifetimes (less than a few minutes) owing to thermal decomposition back to the original reactants. They are thus not effective sinks of NO_x.

AX2.2.2 Nonlinear Relations between Nitrogen Oxide Concentrations and Ozone Formation

Ozone is unlike some other species whose rates of formation vary directly with the emissions of their precursors in that O_3 production ($P(O_3)$) changes nonlinearly with the concentrations of its precursors. At the low NO_x concentrations found in most environments, ranging from remote continental areas to rural and suburban areas downwind of urban centers, the net production of O_3 increases with increasing NO_x. At the high NO_x concentrations found in downtown metropolitan areas, especially near busy streets and roadways, and in power plant plumes, there is net destruction of O_3 by (titration) reaction with NO. Between these two regimes is a transition stage in which O_3 shows only a weak dependence on NO_x concentrations. In the high NO_x regime, NO_2 scavenges OH radicals which would otherwise oxidize VOCs to produce peroxy radicals, which in turn would oxidize NO to NO_2. In the low NO_x regime, VOV (VOC) oxidation generates, or at least does not consume, free radicals, and O_3 production varies directly with NO_x. Sometimes the terms 'VOC-limited' and 'NO_x-limited' are used to describe these two regimes. However, there are difficulties with this usage because: (1) VOC

measurements are not as abundant as they are for NO_x, (2) rate coefficients for reaction of individual VOCs with free radicals vary over an extremely wide range, and (3) consideration is not given to CO nor to reactions that can produce free radicals without invoking VOCs. The terms NO_x-limited and NO_x-saturated (used by, e.g., Jaeglé et al., 2001) will be used wherever possible to describe these two regimes more adequately. However, the terminology used in original articles will also be kept. The chemistry of OH radicals, which are responsible for initiating the oxidation of hydrocarbons, shows behavior similar to that for O_3 with respect to NO_x concentrations (Hameed et al., 1979; Pinto et al., 1993; Poppe et al., 1993; Zimmerman and Poppe, 1993). These considerations introduce a high degree of uncertainty into attempts to relate changes in O_3 concentrations to emissions of precursors. It should also be noted at the outset that in a NO_x-limited (or NO_x-sensitive) regime, O_3 formation is not insensitive to radical production or the flux of solar UV photons, just that O_3 formation is more sensitive to NO_x. For example, global tropospheric O_3 is sensitive to the concentration of CH_4 even though the troposphere is predominantly NO_x-limited.

Various analytical techniques have been proposed that use ambient NO_x and VOC measurements to derive information about O_3 production and O_3-NO_x-VOC sensitivity. Previously (e.g., National Research Council, 1991), it was suggested that O_3 formation in individual urban areas could be understood in terms of measurements of ambient NO_x and VOC concentrations during the early morning. In this approach, the ratio of summed (unweighted by chemical reactivity) VOC to NO_x concentrations is used to determine whether conditions are NO_x-sensitive or VOC sensitive. This technique is inadequate to characterize O_3 formation because it omits many factors recognized as important for $P(O_3)$, including: the effect of biogenic VOCs (which are not present in urban centers during early morning); important individual differences in the ability of VOCs to generate free radicals, rather than just from total VOC concentration and other differences in O_3-forming potential for individual VOCs (Carter, 1995); the effect of multiday transport; and general changes in photochemistry as air moves downwind from urban areas (Milford et al., 1994).

Jacob et al. (1995) used a combination of field measurements and a chemical transport model (CTM) to show that the formation of O_3 changed from NO_x-limited to NO_x-saturated as the season changed from summer to fall at a monitoring site in Shenandoah National Park, VA. Photochemical production of O_3 generally occurs together with production of various other

species including HNO_3, organic nitrates, and hydrogen peroxide (H_2O_2). The relative rates of $P(O_3)$ and the production of other species varies depending on photochemical conditions, and can be used to provide information about O_3-precursor sensitivity.

There are no hard and fast rules governing the levels of NO_x at which the transition from NO_x-limited to NO_x-saturated conditions occurs. The transition between these two regimes is highly spatially and temporally dependent. In the upper troposphere, responses to NO_x additions from commercial aircraft have been found which are very similar to these in the lower troposphere (Bruhl et al., 2000). Bruhl et al. (2000) found that the NO_x levels for O_3 production versus loss are highly sensitive to the radical sources included in model calculations. They found that inclusion of only CH_4 and CO oxidation leads to a decrease in net O_3 production in the North Atlantic flight corridor due to NO emissions from aircraft. However, the additional inclusion of acetone photolysis was found to shift the maximum in O_3 production to higher NO_x mixing ratios, thereby reducing or eliminating areas in which O_3 production rates decreased due to aircraft emissions.

Trainer et al. (1993) suggested that the slope of the regression line between O_3 and summed NO_x oxidation products (NO_z, equal to the difference between measured total reactive nitrogen, NO_y, and NO_x) can be used to estimate the rate of $P(O_3)$ per NO_x (also known as the O_3 production efficiency, or OPE). Ryerson et al. (1998, 2001) used measured correlations between O_3 and NO_z to identify different rates of O_3 production in plumes from large point sources.

Sillman (1995) and Sillman and He (2002) identified several secondary reaction products that show different correlation patterns for NO_x-limited conditions and NO_x-saturated conditions. The most important correlations are for O_3 versus NO_y, O_3 versus NO_z, O_3 versus HNO_3, and H_2O_2 versus HNO_3. The correlations between O_3 and NO_y, and O_3 and NO_z are especially important because measurements of NO_y and NO_x are widely available. Measured O_3 versus NO_z (Figure AX2-2) shows distinctly different patterns in different locations. In rural areas and in urban areas such as Nashville, TN, O_3 shows a strong correlation with NO_z and a relatively steep slope to the regression line. By contrast, in Los Angeles O_3 also increases with NO_z, but the rate of increase of O_3 with NO_z is lower and the O_3 concentrations for a given NO_z value are generally lower.

The difference between NO_x-limited and NO_x-saturated regimes is also reflected in measurements of H_2O_2. Formation of H_2O_2 takes place by self-reaction of photochemically-generated HO_2 radicals, so that there is large seasonal variation of H_2O_2 concentrations, and

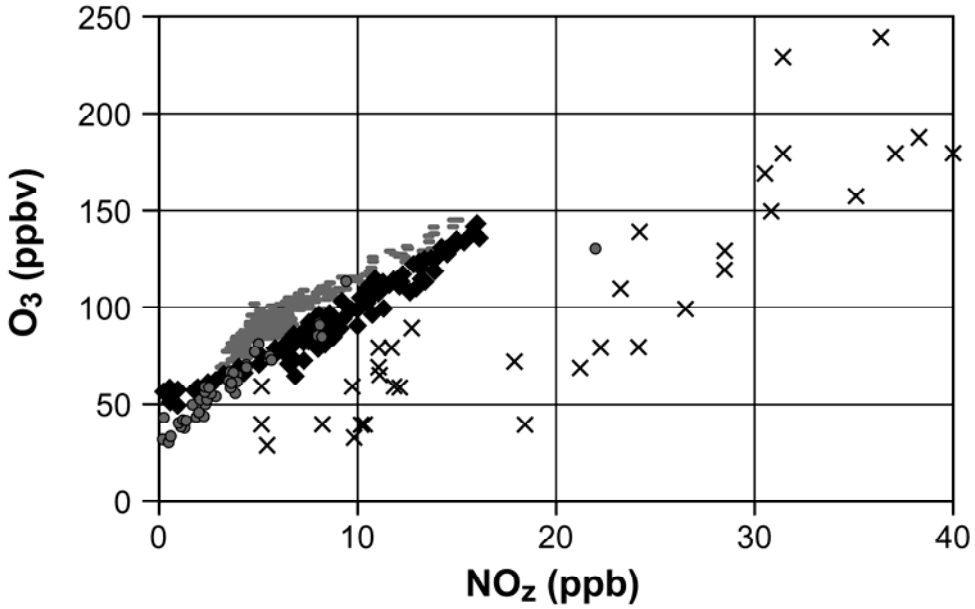


Figure AX2-2. Measured values of O_3 and NO_z (NO_y–NO_x) during the afternoon at rural sites in the eastern United States (gray circles) and in urban areas and urban plumes associated with Nashville, TN (gray dashes), Paris, FR (black diamonds) and Los Angeles, CA (X's)

values in excess of 1 ppb are mainly limited to the summer months when photochemistry is more active (Kleinman, 1991). Hydrogen peroxide is produced in abundance only when O_3 is produced under NO_x-limited conditions. When the photochemistry is NO_x-saturated, much less H_2O_2 is produced. In addition, increasing NO_x tends to slow the formation of H_2O_2 under NO_x-limited conditions. Differences between these two regimes are also related to the preferential formation of sulfate during summer and to the inhibition of sulfate and hydrogen peroxide during winter (Stein and Lamb, 2003). Measurements in the rural eastern United States (Jacob et al., 1995), at Nashville (Sillman et al., 1998), and at Los Angeles (Sakugawa and Kaplan, 1989) show large differences in H_2O_2 concentrations likely due to differences in NO_x availability at these locations.

AX2.2.3 Multiphase Chemistry Involving NO_x

Recent laboratory studies on sulfate and organic aerosols indicate that the reaction probability γ_{N2O5} is in the range of 0.01 to 0.05 (Kane et al., 2001; Hallquist et al., 2003; Thornton et al., 2003). Tie et al. (2003) found that a value of 0.04 in their global model gave the best simulation of observed NO_x concentrations over the Arctic in winter.

Using aircraft measurements over the northeastern U.S., Brown et al. (2006b) found that the uptake coefficient for N_2O_5, γN_2O_5, on the surfaces of particles depends strongly on their sulfate content. They found that γN_2O_5 was highest (0.017) in regions where the aerosol sulfate concentration was highest and lower elsewhere (<0.0016). This result contrasts with that of Dentener and Crutzen (1993) who concluded that γN_2O_5 would be independent of aerosol composition, based on a value for γN_2O_5 of 0.1, implying that the heterogeneous hydrolysis of N_2O_5 would be saturated for typical ambient aerosol surface areas. The importance of this reaction to tropospheric chemistry depends on the value of γN_2O_5. If it is 0.01 or lower, there may be difficulty in explaining the loss of NO_y and the formation of aerosol nitrate, especially during winter. A decrease in N_2O_5 slows down the removal of NO_x by leaving more NO_2 available for reaction and thus increases O_3 production. Based on the consistency between measurements of NO_y partitioning and gas-phase models, Jacob (2000) considers it unlikely that HNO_3 is recycled to NO_x in the lower troposphere in significant concentrations. However, only one of the reviewed studies (Schultz et al., 2000) was conducted in the marine troposphere and none was conducted in the MBL. An investigation over the equatorial Pacific reported discrepancies between observations and theory (Singh et al., 1996) which might be explained by HNO_3 recycling. It is important to recognize that both Schultz et al. (2000) and Singh et al. (1996) involved aircraft sampling at altitude which, in the MBL, can significantly under-represent sea salt aerosols and thus most total NO_3 (defined to be $HNO_3 + NO_3^-$) and large fractions of NO_y in marine air (e.g., Huebert et al., 1996). Consequently, some caution is warranted when interpreting constituent ratios and NO_y budgets based on such data.

Recent work in the Arctic has quantified significant photochemical recycling of NO_3^- to NO_x and attendant perturbations of OH chemistry in snow (Honrath et al., 2000; Dibb et al., 2002; Domine and Shepson, 2002) which suggest the possibility that similar multiphase pathways could occur in aerosols. As mentioned above, NO_3^- is photolytically reduced to NO_2^- (Zafiriou and True, 1979) in acidic sea salt solutions (Anastasio et al., 1999). Further photolytic

reduction of NO_2^- to NO (Zafariou and True, 1979) could provide a possible mechanism for HNO_3 recycling. Early experiments reported production of NO_x during the irradiation of artificial seawater concentrates containing NO_3^- (Petriconi and Papee, 1972). Based on the above, HNO_3 recycling in sea salt aerosols is potentially important and warrants further investigation. Other possible recycling pathways involving highly acidic aerosol solutions and soot are reviewed by Jacob (2000).

Stemmler et al. (2006) studied the photosensitized reduction of NO_2 to HONO on humic acid films using radiation in the UV-A through the visible spectral regions. They also found evidence for reduction occurring in the dark, reactions which may occur involving surfaces containing partly oxidized aromatic structures. For example, Simpson et al. (2006) found that aromatic compounds constituted ~20% of organic films coating windows in downtown Toronto. They calculated production rates of HONO that are compatible with observations of high HONO levels in a variety of environments. The photolysis of HONO formed this way could account for up to 60% of the integrated source of OH radicals in the inner planetary boundary layer. A combination of high NO_2 levels and surfaces of soil and buildings and other man-made structures exposed to diesel exhaust would then be conducive to HONO formation and, hence, to high [OH] (Xu et al., 2006).

Ammann et al. (1998) reported the efficient conversion of NO_2 to HONO on fresh soot particles in the presence of water. They suggest that interaction between NO_2 and soot particles may account for high mixing ratios of HONO observed in urban environments. Conversion of NO_2 to HONO and subsequent photolysis and HONO to NO + OH would constitute a NO_x-catalyzed O_3 sink involving snow. High concentrations of HONO can lead to the rapid growth in OH concentrations shortly after sunrise, giving a "jump start" to photochemical smog formation. Prolonged exposure to ambient oxidizing agents appears to deactivate this process. Broske et al. (2003) studied the interaction of NO_2 on secondary organic aerosols and concluded that the uptake coefficients were too low for this reaction to be an important source of HONO in the troposphere.

Choi and Leu (1998) evaluated the interactions of HNO_3 on model black carbon soot (FW2), graphite, hexane, and kerosene soot. They found that HNO_3 decomposed to NO_2 and H_2O at higher HNO_3 surface coverages, i.e., $P(HNO_3) >= 10^{-4}$ Torr. None of the soot models used were reactive at low HNO_3 coverages, at $P(HNO_3) = 5 \times 10^{-7}$ Torr or at temperatures below

220 K. They conclude that it is unlikely that aircraft soot in the upper troposphere/lower stratosphere reduces HNO_3.

Heterogeneous production on soot at night is believed to be the mechanism by which HONO accumulates to provide an early morning source of HO_x in high NO_x environments (Harrison et al., 1996; Jacob, 2000). HONO has been frequently observed to accumulate to levels of several ppb overnight, and this has been attributed to soot chemistry (Harris et al., 1982; Calvert et al., 1994; Jacob, 2000).

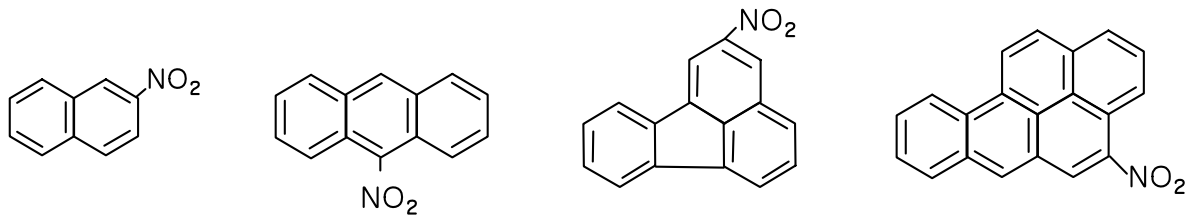
Longfellow et al. (1999) observed the formation of HONO when methane, propane, hexane, and kerosene soots were exposed to NO_2. They suggested that this reaction may account for some part of the unexplained high levels of HONO observed in urban areas. They comment that without details about the surface area, porosity, and amount of soot available for this reaction, reactive uptake values cannot be estimated reliably. They comment that soot and NO_2 are produced in close proximity during combustion, and that large quantities of HONO have been observed in aircraft plumes.

Saathoff et al. (2001) studied the heterogeneous loss of NO_2, HNO_3, NO_3/N_2O_5, HO_2/HO_2NO_2 on soot aerosol using a large aerosol chamber. Reaction periods of up to several days were monitored and results used to fit a detailed model. Saathoff et al. derived reaction probabilities at 294 K and 50% RH for NO_2, NO_3, HO_2, and HO_2NO_2 deposition to soot; HNO_3 reduction to NO_2; and N_2O_5 hydrolysis. When these probabilities were included in photochemical box model calculations of a 4-day smog event, the only noteworthy influence of soot was a 10% reduction in the second day O_3 maximum, for a soot loading of 20 $\mu g\ m^{-3}$, i.e., roughly a factor of 10 times observed black carbon loadings seen in U.S. urban areas, even during air pollution episodes.

Muñoz and Rossi (2002) conducted Knudsen cell studies of HNO_3 uptake on black and grey decane soot produced in lean and rich flames, respectively. They observed HONO as the main species released following HNO_3 uptake on grey soot, and NO and traces of NO_2 from black soot. They conclude that these reactions would only have relevance in special situations in urban settings where soot and HNO_3 are present in high concentrations simultaneously.

Formation of Nitro PAHs

Nitro-polycyclic aromatic hydrocarbons (nitro-PAHs) (see Figure AX2-3 for some example nitro-PAHs) are generated from incomplete combustion processes through electrophilic



2-nitronaphthalene 9-nitroanthracene 2-nitrofluoranthene 6-nitrobenzo(a)pyrene

Figure AX2-3. Structures of nitro-polycyclic aromatic hydrocarbons.

reactions of polycyclic aromatic hydrocarbons (PAHs) in the presence of NO_2 (International Agency for Research on Cancer [IARC], 1989; World Health Organization [WHO], 2003). Among combustion sources, diesel emissions have been identified as the major source of nitro-PAHs in ambient air (Bezabeh et al., 2003; Gibson, 1983; Schuetzle, 1983; Tokiwa and Ohnishi, 1986). Direct emissions of NPAHs in PM vary with type of fuel, vehicle maintenance, and ambient conditions (Zielinska et al., 2004).

In addition to being directly emitted, nitro-PAHs can also be formed from both gaseous and heterogeneous reactions of PAHs with gaseous nitrogenous pollutants in the atmosphere (Arey et al., 1986; Arey et al., 1989, Arey, 1998; Perrini, 2005; Pitts, 1987; Sasaki et al., 1997; Zielinska et al., 1989). Different isomers of nitro-PAHs are formed through different formation processes. For example, the most abundant nitro-PAH in diesel particles is 1-nitropyene (1NP), followed by 3-nitrofluoranthene (3NF) and 8-nitrofluoranthene (8NF) (Bezabeh et al., 2003; Gibson, 1983; Schuetzle, 1983; Tokiwa and Ohnishi, 1986). However, in ambient particulate organic matter (POM), 2-nitrofluoranthene (2NF) is the dominant compound, followed by 1NP and 2-nitropyrene (2NP) (Arey et al., 1989; Bamford et al., 2003; Reisen and Arey, 2005; Zielinska et al., 1989), although 2NF and 2NP are not directly emitted from primary combustion emissions. The reaction mechanisms for the different nitro-PAH formation processes have been well documented and are presented in Figure AX2-3.

The dominant process for the formation of nitro-PAHs in the atmosphere is gas-phase reaction of PAHs with OH radicals in the presence of NO_x (Arey et al., 1986, Arey, 1998; Atkinson and Arey, 1994; Ramdahl et al., 1986; Sasaki et al., 1997). Hydroxyl radicals can be generated photochemically or at night through ozone-alkene reactions, (Finlayson-Pitts and Pitts,

2000). The postulated reaction mechanism of OH with PAHs involves the addition of OH at the site of highest electron density of the aromatic ring, for example, the 1-position for pyrene (PY) and the 3-position for fluoranthene (FL). This reaction is followed by the addition of NO_2 to the OH-PAH adduct and elimination of water to form the nitroarenes (Figure AX2-4, Arey et al., 1986; Aktinson et al., 1990; Pitts, 1987). After formation, nitro-PAHs with low vapor pressures (such as 2NF and 2NP) immediately migrate to particles under ambient conditions (Fan et al., 1995; Feilberg et al., 1999). The second order rate-constants for the reactions of OH with most PAHs range from 10^{-10} to 10^{-12} $cm^3molecule^{-1}s^{-1}$ at 298 K with the yields ranging from ~0.06 to ~5% (Atkinson and Arey, 1994). 2NF and 2NP have been found as the most abundant nitro-PAHs formed via reactions of OH with gaseous PY and FL, respectively in ambient air.

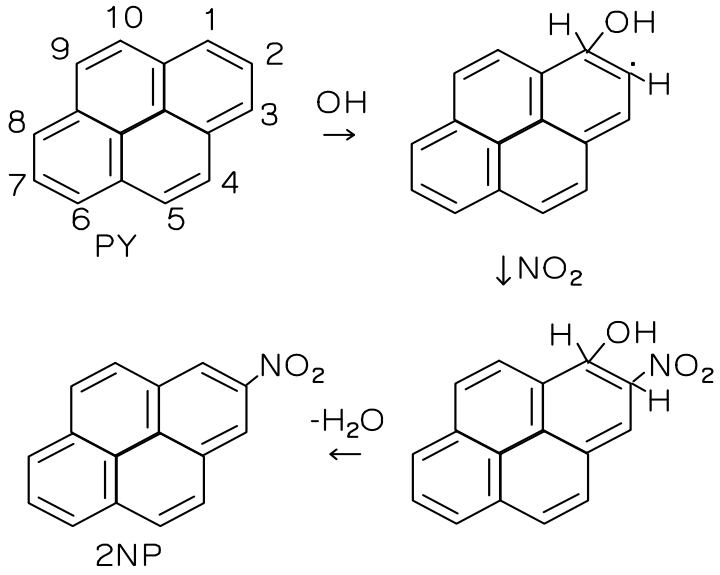


Figure AX2-4. Formation of 2-nitropyrene (2NP) from the reaction of OH with gaseous pyrene (PY).

The second important process for the formation of nitro-PAHs in the atmosphere is the nitration of PAHs by NO_3^- in the presence of NO_x at night (Atkinson et al., 1990; Atkinson and Arey, 1994; Sasaki et al., 1997). Nitrate radicals can be generated by reaction of ozone (O_3) with NO_2 in the atmosphere by Reaction AX2-5:

$$O_3 + NO_2 \rightarrow NO_3 + O_2 \quad \text{(AX2-5)}$$

Similar to the mechanism of OH reactions with PAHs, NO_3 initially adds to the PAH ring to form an NO_3-PAH adduct, followed by loss of HNO_3 to form nitro-PAHs (Atkinson et al., 1990; Atkinson and Arey, 1994; Sasaki et al., 1997). For example, in the mixture of naphthalene and N_2O_5-NO_3-NO_2, the major products formed through the NO_3 reaction are 1- and 2-nitro-naphthalene (1NN and 2NN) (Atkinson et al., 1990; Feilberg et al., 1999; Sasaki et al., 1997). 2NF and 4NP were reported as the primary products of the gas-phase reactions of FL and PY with NO_3 radical, respectively (Atkinson et al., 1990; Atkinson and Arey, 1994).

The reaction with NO_3 is of minor importance in the daytime because NO_3 radical is not stable in sunlight. In addition, given the rapid reactions of NO with NO_3 and with O_3 in the atmosphere (Finlayson-Pitts and Pitts 2000), concentrations of NO_3 at ground level are low during daytime. However, at night, concentrations of NO_3 radicals formed in polluted ambient air are expected to increase. According to Atkinson et al. (1991), the average NO_3 concentration is about 20 ppt in the lower troposphere at night and can be as high as 430 ppt. It is also worth noting that significant NO_3 radical concentrations are found at elevated altitudes where O_3 is high but NO is low (Reissell and Arey, 2001; Stutz et al., 2004). When NO_3 reaches high concentrations, the formation of nitro-PAHs by the reaction of gaseous PAHs with NO_3 may be of environmental significance. At $10^{-17} - 10^{-18}$ cm^3 $molecule^{-1}s^{-1}$, the rate constants of NO_3 with most PAHs are several orders of magnitude lower than those of OH with the same PAHs; however, the yields of nitro-PAHs from NO_3 reactions are generally much higher than those of OH reactions. For example, the yields of 1-NN and 2NF are 0.3% and 3%, respectively from OH reactions, but the yields are 17% and 24% for these two compounds generated from the NO_3 radical reactions (Atkinson and Arey 1994). Therefore, formation of nitro-PAHs via reactions of NO_3 at nighttime under certain circumstances can be significant.

The third process of nitro-PAH formation in the atmosphere is nitration of PAHs by NO_2/N_2O_5 in the presence of trace amounts of HNO_3 (HNO_3) in both gas and particle phases. This mechanism could be operative throughout the day and night (Pitts et al., 1983, 1985a, b; Grosjean et al., 1983; Ramdahl et al., 1984; Kamens et al., 1990). The formation of nitro-fluoranthenes was observed when adsorbed FL was exposed to gaseous N_2O_5, and the distribution of product NF isomers was 3- > 8- > 7- > 1- NF (Pitts et al., 1985a, b). The

proposed mechanism for this reaction was an ionic electrophilic nitration by nitronium ion (NO_2^+). It was speculated that N_2O_5 became ionized prior to the reaction with FL (Zielinska et al., 1986). Only 1NP was observed for the reaction of PY with N_2O_5 on filters (Pitts et al., 1985b). Compared to the reactions of OH and NO_3, nitration of PAHs by NO_2/N_2O_5 is less important.

Measurements of nitro-PAHs in ambient air provide evidence for the proposed reaction mechanism, i.e. the reactions of OH and NO_3 radicals with PAHs are the major sources of nitro-PAHs (Bamford and Baker, 2003; Reisen and Arey, 2005; and references therein). 2NF is a ubiquitous component of ambient POM, much higher than 1NP, itself a marker of combustion sources. Nitro-PAH isomer ratios show strong seasonality. For instance, the mean ratios of 2NF/1NP were higher in summer than in winter (Bamford et al., 2003; Reisen and Arey, 2005), indicating that reactions of OH and NO_3 with FL are the major sources of nitro-PAHs in ambient air in summer. The ratio of 2NF/1NP was lower in winter than in summer because of lower OH concentrations and, therefore, less production of 2NF via atmospheric reactions. A ratio of 1NP/2NF greater than 1 was observed in locations with major contributions from vehicle emissions (Dimashki et al., 2000; Feilberg et al., 2001). In addition, the ratio of 2NF/2NP was also used to evaluate the contribution of OH and NO_3 initiated reactions to the ambient nitro-PAHs (Bamford et al., 2003; Reisen and Arey, 2005).

The concentrations for most nitro-PAHs found in ambient air are much lower than 1 pg/m^3, except NNs, 1NP, and 2NF, which can be present at several pg/m^3. These levels are much lower (~2 to ~1000 times lower) than their parent PAHs. However, nitro-PAHs are much more toxic than PAHs (Durant et al., 1996; Grosovsky et al., 1999; Salmeen et al., 1982; Tokiwa et al., 1998; Tokiwa and Ohnishi, 1986). Moreover, most nitro-PAHs are present in particles with a mass median diameter <0.1 μm.

Esteve et al. (2006) examined the reaction of gas-phase NO_2 and OH radicals with various PAHs adsorbed onto model diesel particulate matter (SRM 1650a, NIST). Using pseudo second order rate coefficients, they derived lifetimes for conversion of the particle-bound PAHs to nitro-PAHs of a few days (for typical urban NO_2 levels of 20 ppb). They also found that the rates of reaction of OH with the PAHs were about four orders of magnitude larger than for the reactions involving NO_2. However, since the concentrations of NO_2 used above are more than four orders of magnitude larger than those for OH (10^6-10^7/cm^3), they concluded that the

pathway involving NO_2 is expected to be favored over that involving OH radicals. Consistent with the importance of the gas-phase formation of NPAHS, both the mutagenic potency of PM and the content of NPAHs in PM vary by particle size, and are higher in the submicron size range (Xu and Lee, 2000; Kawanaka et al., 2004).

The major loss process of nitro-PAHs is photodecomposition (Fan et al., 1996; Feilberg et al., 1999; Feilberg and Nielsen, 2001), with lifetimes on the order of hours. However, lacking direct UV light sources indoors, nitro-PAHs are expected have a longer lifetimes (days) indoors than outdoors; and may therefore pose increased health risks. Many nitro-PAHs are semi- or nonvolatile organic compounds. As stated above, indoor environments have much greater surface areas than outdoors. Thus, it is expected that gas/particle distribution of nitro-PAHs indoors will be different from those in ambient air. A significant portion of nitro-PAHs will probably be adsorbed by indoor surfaces, such as carpets, leading to different potential exposure pathways to nitro-PAHs in indoor environments. The special characteristics of indoor environments, which can affect the indoor chemistry and potential exposure pathways significantly, should be taken into consideration when conducting exposure studies of nitro-PAHs.

Reaction with OH and NO_3 radicals is a major mechanism for removing gas-phase PAHs, with OH radical initiated reactions predominating depending on season (Vione et al., 2004; Bamford et al., 2003). Particle-bound PAH reactions occur but tend to be slower. Nitronaphthalenes tend to remain in the vapor phase, but because phase partitioning depends on ambient temperature, in very cold regions these species can condense (Castells et al., 2003) while the higher molecular weight PAHs such as the nitroanthracenes, nitrophenantrenes and nitrofluoranthenes condense in and on PM (Ciganek et al., 2004; Cecinato, 2003).

Multiphase Chemical Processes Involving Nitrogen Oxides and Halogens

Four decades of observational data on O_3 in the troposphere have revealed numerous anomalies not easily explained by gas-phase HO_x-NO_x photochemistry. The best-known example is the dramatic depletion of ground-level O_3 during polar sunrise due to multiphase catalytic cycles involving inorganic Br and Cl radicals (Barrie et al., 1988; Martinez et al., 1999; Foster et al., 2001). Other examples of anomalies in tropospheric O_3 at lower latitudes include low levels of O_3 (<10 ppbv) in the marine boundary layer (MBL) and overlying free troposphere (FT) at times over large portions of the tropical Pacific (Kley et al., 1996), as well as post-sunrise

O_3 depletions over the western subtropical Pacific Ocean (Nagao et al., 1999), the temperate Southern Ocean (Galbally et al., 2000), and the tropical Indian Ocean (Dickerson et al., 1999). The observed O_3 depletions in near-surface marine air are generally consistent with the model-predicted volatilization of Br_2, BrCl, and Cl_2 from sea salt aerosols through autocatalytic halogen "activation" mechanisms (e.g., Vogt et al., 1996; von Glasow et al., 2002a) involving these aqueous phase reactions.

$$HOBr + Br^- + H^+ \rightarrow Br_2 + H_2O \quad \text{(AX2-23)}$$

$$HOCL + Br^- + H^+ \rightarrow BrC1 + H_2O \quad \text{(AX2-24)}$$

$$HOC1 + C1^- + H^+ \rightarrow C1_2 + H_2O \quad \text{(AX2-25)}$$

In polluted marine regions at night, the heterogeneous reaction

$$N_2O_5 + C1^- \rightarrow C1NO_2 + NO_3^- \quad \text{(AX2-26)}$$

may also be important (Finlayson-Pitts et al., 1989; Behnke et al., 1997; Erickson et al., 1999). Diatomic bromine, BrCl, Cl_2, and $ClNO_2$ volatilize and photolyze in sunlight to produce atomic Br and Cl. The acidification of sea salt aerosol via incorporation of HNO_3 (and other acids) leads to the volatilization of HCl (Erickson et al., 1999), e.g.

$$HNO_3 + C1^- \rightarrow HC1 + NO_3^- \quad \text{(AX2-27)}$$

and the corresponding shift in phase partitioning can accelerate the deposition flux to the surface of total NO_3 (Russell et al., 2003; Fischer et al., 2006). However, Pryor and Sorensen (2000) have shown that the dominant form of nitrate deposition is a complex function of wind speed. In polluted coastal regions where HCl from Reaction 35 often exceeds 1 ppbv, significant additional atomic Cl^- is produced via:

$$HC1 + OH \rightarrow C1 + H_2O \quad \text{(AX2-28)}$$

(Singh and Kasting, 1988; Keene et al., 2007). Following production, Br and Cl atoms catalytically destroy O_3 via:

$$X + O_3 \rightarrow XO + O_2 \quad \text{(AX2-29)}$$

$$XO + HO_2 \rightarrow HOX + O_2 \quad \text{(AX2-30)}$$

$$HOX + h\nu \rightarrow OH + X \quad \text{(AX2-31)}$$

where (X = Br and Cl).

Formation of Br and Cl nitrates via

$$XO + NO_2 \rightarrow XNO_3 \quad \text{(AX2-32)}$$

and the subsequent reaction of XNO_3 with sea salt and sulfate aerosols via

$$XNO_3 + H_2O \rightarrow HOX + H^+ + NO_3^- \quad \text{(AX2-33)}$$

and:

$$XNO_3 + Y^- \rightarrow XY + NO_3^- \quad \text{(AX2-34)}$$

(where Y = Cl, Br, or I) accelerates the conversion of NO_x to particulate NO_3^- and thereby contributes indirectly to net O_3 destruction (Sander et al., 1999; Vogt et al., 1999, Pszenny et al., 2004). Most XNO_3 reacts via reaction 34 on sea salt whereas reaction 33 is more important on sulfate aerosols. Partitioning of HCl on sulfate aerosols following Henry's Law provides Cl^- for reaction 34 to form BrCl. Product NO_3^- from both reactions AX2-33 and AX2-34 partitions with the gas-phase HNO_3 following Henry's Law. Because most aerosol size fractions in the MBL are near equilibrium with respect to HNO_3 (Erickson et al., 1999; Keene et al., 2004), both sulfate and sea salt aerosol can sustain the catalytic removal of NO_x and re-activation of Cl and Br with no detectable change in composition. The photolytic reduction of NO_3^- in sea salt aerosol solutions recycles NO_x to the gas phase (Pszenny et al., 2004). Halogen chemistry also impacts O_3 indirectly by altering OH/HO_2 ratios ($XO + HO_2 \rightarrow HOX + O_2 \rightarrow OH + X$) (e.g., Stutz et al., 1999; Bloss et al., 2005).

In addition to O_3 destruction via reaction AX2-37, atomic Cl oxidizes hydrocarbons (HCs) primarily via hydrogen abstraction to form HCl vapor and organz products (Jobson et al., 1994; Pszenny et al., 2006). The enhanced supply of odd-H radicals from HC oxidation leads to

net O_3 production in the presence of sufficient NO_x (Pszenny et al., 1993). Available evidence suggests that Cl^- radical chemistry may be a significant net source for O_3 in polluted coastal/urban air (e.g., Tanaka et al., 2003; Finley and Saltzman, 2006).

An analogous autocatalyic O_3 destruction cycle involving multiphase iodine chemistry also operates in the marine atmosphere (Alicke et al., 1999, Vogt et al., 1999; McFiggans et al., 2000; Ashworth et al., 2002). In this case, the primary source of I is believed to be either photolysis of CH_2I_2, other I-containing gases (Carpenter et al., 1999; Carpenter, 2003), and/or perhaps I_2 (McFiggans et al., 2004; Saiz-Lopez and Plane, 2004; McFiggans, 2005) emitted by micro-and macro flora. Sea salt and sulfate aerosols provide substrates for multiphase reactions that sustain the catalytic I-IO cycle. The IO radical has been measured by long-path (LP) and/or multi axis (MAX) differential optical absorption spectroscopy (DOAS) at Mace Head, Ireland; Tenerife, Canary Islands; Cape Grim, Tasmania; and coastal New England, USA; having average daytime levels of about 1 ppt with maxima up to 7 ppt (e.g., Allan et al., 2000; Pikelnaya et al., 2006). Modeling suggests that up to 13% per day of O_3 in marine air may be destroyed via multiphase iodine chemistry (McFiggans et al., 2000). The reaction of IO with NO_2 followed by uptake of INO_3 into aerosols (analogous to Reactions AX2-9 through AX2-11) accelerates the conversion of NO_x to particulate NO_3^- and thereby contributes to net O_3 destruction. The reaction $IO + NO \rightarrow I + NO_2$ also influences NO_x cycling.

Most of the above studies have focused on halogen-radical chemistry and related influences on NO_x cycling in coastal and urban air. However, available evidence suggests that similar chemical transformations proceed in other halogen-rich tropospheric regimes. For example, Cl, Br, and/or I oxides have been measured at significant concentrations in near-surface air over the Dead Sea, Israel, the Great Salt Lake, Utah (e.g., Hebestreit et al., 1999; Stutz et al., 1999, 2002; Zingler and Platt, 2005), and the Salar de Uyuni salt pan in the Andes mountains (U. Platt, unpublished data, 2006); high column densities of halogenated compounds have also been observed from satellites over the northern Caspian Sea (Wagner et al., 2001; Hollwedel et al., 2004). The primary source of reactive halogens in these regions is thought to be from activation along the lives of that in reactions in AX2-23 through AX2-25 involving concentrated salt deposits on surface evaporite pans. High concentrations of BrO have also been measured in volcanic plumes (Bobrowski et al., 2003, Gerlach, 2004). Although virtually unexplored, the substantial emissions of inorganic halogens during biomass burning (Lobert et al., 1999; Keene

et al., 2006) and in association with crustal dust (Keene et al., 1999; Sander et al., 2003) may also support active halogen-radical chemistry and related transformations involving NO_x downwind of sources. Finally, observations from satellites, balloons, and aircraft indicate that BrO is present in the free troposphere at levels sufficient to significantly influence photochemistry (e.g., von Glasow et al., 2004).

AX2.3 CHEMISTRY OF SULFUR OXIDES IN THE TROPOSPHERE

The four known monomeric sulfur oxides are sulfur monoxide (SO), sulfur dioxide (SO_2), sulfur trioxide (SO_3), and disulfur monoxide (S_2O). SO can be formed by photolysis of SO_2 at wavelengths less than 220 nm, and so could only be found in the middle and upper stratosphere (Pinto et al., 1989). SO_3 can be emitted from the stacks of power plants and factories however, it reacts extremely rapidly with H_2O in the stacks or immediately after release into the atmosphere to form H_2SO_4. Of the four species, only SO_2 is present at concentrations significant for atmospheric chemistry and human exposures.

Sulfur dioxide can be oxidized either in the gas phase, or, because it is soluble, in the aqueous phase in cloud drops. The gas-phase oxidation of SO_2 proceeds through the reaction

$$SO_2 + OH + M \rightarrow HSO_3 + M \quad \text{(AX2-35)}$$

followed by

$$HSO_3 + O_2 \rightarrow SO_3 + HO_2 \quad \text{(AX2-36)}$$

$$SO_3 + H_2O \rightarrow H_2SO_4 \quad \text{(AX2-37)}$$

Since H_2SO_4 is extremely soluble, it will be removed rapidly by transfer to the aqueous phase of aerosol particles and cloud drops. Rate coefficients for reaction of SO_2 with HO_2 or NO_3 are too low to be significant (JPL, 2003).

SO_2 is chiefly but not exclusively primary in origin; it is also produced by the photochemical oxidation of reduced sulfur compounds such as dimethyl sulfide (CH_3-S-CH_3), hydrogen sulfide (H_2S), carbon disulfide (CS_2), carbonyl sulfide (OCS), methyl mercaptan (CH_3-S-H), and dimethyl disulfide (CH_3-S-S-CH_3) which are all mainly biogenic in origin.

Their sources are discussed in Section AX2.5. Table AX2-1 lists the atmospheric lifetimes of reduced sulfur species with respect to reaction with various oxidants. Except for OCS, which is lost mainly by photolysis (τ~6 months), all of these species are lost mainly by reaction with OH and NO_3 radicals. Because OCS is relatively long-lived in the troposphere, it can be transported upwards into the stratosphere. Crutzen (1976) proposed that its oxidation serves as the major source of sulfate in the stratospheric aerosol layer sometimes referred to the "Junge layer," (Junge et al., 1961) during periods when volcanic plumes do not reach the stratosphere. However, the flux of OCS into the stratosphere is probably not sufficient to maintain this stratospheric aerosol layer. Myhre et al. (2004) propose instead that SO_2 transported upwards from the troposphere is the most likely source, have become the upward flux of OCS is too small to sustain observed sulfate loadings in the Junge layer. In addition, insitu measurements of the isotopic composition of sulfur do not match those of OCS (Leung et al., 2002). Reaction with NO_3 radicals at night most likely represents the major loss process for dimethyl sulfide and methyl mercaptan. The mechanisms for the oxidation of DMS are still not completely understood. Initial attack by NO_3 and OH radicals involves H atom abstraction, with a smaller branch leading to OH addition to the S atom. The OH addition branch increases in importance as temperatures decrease and becoming the major pathway below temperatures of 285 K (Ravishankara, 1997). The adduct may either decompose to form methane sulfonic acid (MSA), or undergo further reactions in the main pathway, to yield dimethyl sulfoxide (Barnes et al., 1991). Following H atom abstraction from DMS, the main reaction products include MSA and SO_2. The ratio of MSA to SO_2 is strongly temperature dependent, varying from about 0.1 in tropical waters to about 0.4 in Antarctic waters (Seinfeld and Pandis, 1998). Excess sulfate (over that expected from the sulfate in seawater) in marine aerosol is related mainly to the production of SO_2 from the oxidation of DMS. Transformations among atmospheric sulfur compounds are summarized in Figure AX2-5.

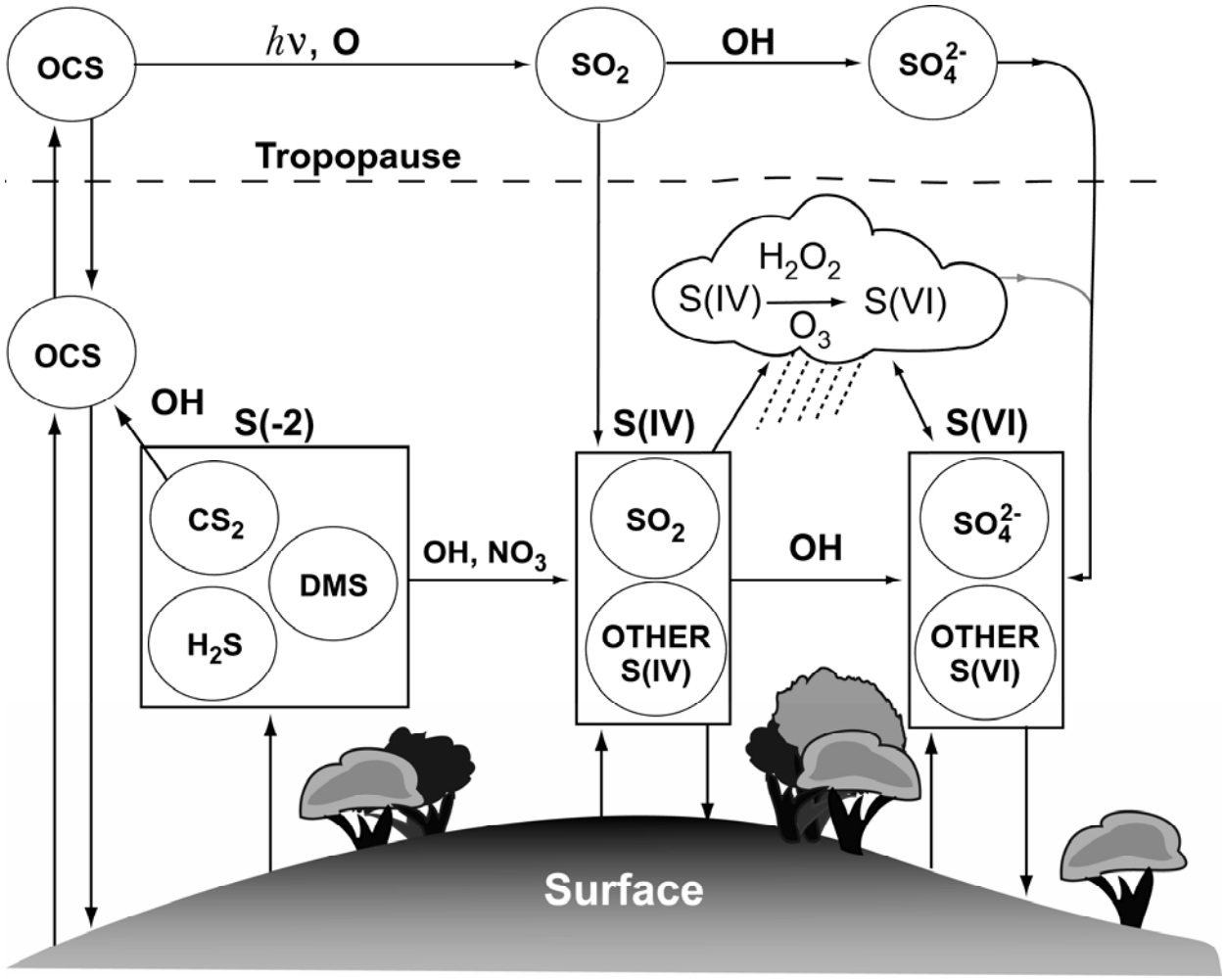


Figure AX2-5. Transformations of sulfur compounds in the atmosphere.

Source: Adapted from Berresheim et al. (1995).

Multiphase Chemical Processes Involving Sulfur Oxides and Halogens

Chemical transformations involving inorganic halogenated compounds effect changes in the multiphase cycling of sulfur oxides in ways analogous to their effects on NO_x. Oxidation of dimethylsulfide $(CH_3)_2S$ by BrO produces dimethylsulfoxide $(CH_3)_2SO$ (Barnes et al., 1991; Toumi, 1994), and oxidation by atomic chloride leads to formation of SO_2 (Keene et al., 1996). $(CH_3)_2SO$ and SO_2 are precursors for methanesulfonic acid (CH_3SO_3H) and H_2SO_4. In the MBL, virtually all H_2SO_4 and CH_3SO_3H vapor condenses onto existing aerosols or cloud droplet, which subsequently evaporate, thereby contributing to aerosol growth and acidification. Unlike CH_3SO_3H, H_2SO_4 also has the potential to produce new particles (Korhonen et al., 1999; Kumala

et al., 2000), which in marine regions is thought to occur primarily in the free troposphere. Saiz-Lopez et al. (2004) estimated that observed levels of BrO at Mace Head would oxidize $(CH_3)_2S$ about six times faster than OH and thereby substantially increase production rates of H_2SO_4 and other condensible S species in the MBL. Sulfur dioxide is also scavenged by deliquesced aerosols and oxidized to H_2SO_4 in the aqueous phase by several strongly pH-dependent pathways (Chameides and Stelson, 1992; Vogt et al., 1996; Keene et al., 1998). Model calculations indicate that oxidation of S(IV) by O_3 dominates in fresh, alkaline sea salt aerosols, whereas oxidation by hypohalous acids (primarily HOCl) dominates in moderately acidic solutions. Additional particulate non-sea salt (nss) SO_4^{2-} is generated by SO_2 oxidation in cloud droplets (Clegg and Toumi, 1998). Ion-balance calculations indicate that most nss SO_4^{2-} in short-lived (two to 48 hours) sea salt size fractions accumulates in acidic aerosol solutions and/or in acidic aerosols processed through clouds (e.g., Keene et al., 2004). The production, cycling, and associated radiative effects of S-containing aerosols in marine and coastal air are regulated in part by chemical transformations involving inorganic halogens (von Glasow et al., 2002b). These transformations include: dry-deposition fluxes of nss SO_4^{2-} in marine air dominated, naturally, by the sea salt size fractions (Huebert et al., 1996; Turekian et al., 2001); HCl phase partitioning that regulates sea salt pH and associated pH-dependent pathways for S(IV) oxidation (Keene et al., 2002; Pszenny et al., 2004); and potentially important oxidative reactions with reactive halogens for $(CH_3)2S$ and S(IV). However, both the absolute magnitudes and relative importance of these processes in MBL S cycling are poorly understood.

Iodine chemistry has been linked to ultrafine particle bursts at Mace Head (O'Dowd et al., 1999, 2002). Observed bursts coincide with the elevated concentrations of IO and are characterized by particle concentrations increasing from background levels to up to 300,000 cm^{-3} on a time scale of seconds to minutes. This newly identified source of marine aerosol would provide additional aerosol surface area for condensation of sulfur oxides and thereby presumably diminish the potential for nucleation pathways involving H_2SO_4. However, a subsequent investigation in polluted air along the New England, USA coast found no correlation between periods of nanoparticle growth and corresponding concentrations of I oxides (Russell et al., 2006). The potential importance of I chemistry in aerosol nucleation and its associated influence on sulfur cycling remain highly uncertain.

AX2.4 MECHANISMS FOR THE AQUEOUS PHASE FORMATION OF SULFATE AND NITRATE

The major species containing sulfur in clouds are HSO_3^- and SO_3^{2-}, which are derived from the dissolution of SO_2 in water and are referred to as S(IV); and HSO_4^- and SO_4^{2-}, which are referred to as S(VI). The major species capable of oxidizing S(IV) to S(VI) in cloud water are O_3, peroxides (either H_2O_2 or organic peroxides), OH radicals, and ions of transition metals such as Fe and Cu that can catalyze the oxidation of S(IV) to S(VI) by O_2.

The basic mechanism of the aqueous phase oxidation of SO_2 has long been studied and can be found in numerous texts on atmospheric chemistry, e.g., Seinfeld and Pandis (1998), Jacob (2000), and Jacobson (2002). The steps involved in the aqueous phase oxidation of SO_2 can be summarized as follows (Jacobson, 2002):

Dissolution of SO_2

$$SO_2(g) \Leftrightarrow SO_2(aq) \qquad \text{(AX2-38)}$$

The formation and dissociation of H_2SO_3

$$SO_2(aq) + H_2O(aq) \Leftrightarrow H_2SO_3 \Leftrightarrow H^+ + HSO_3^- \Leftrightarrow 2H^+ + SO_3^{2-} \qquad \text{(AX2-39)}$$

In the pH range commonly found in rainwater (2 to 6), the most important reaction converting S(IV) to S(VI) is

$$HSO_3^- + H_2O_2 + H^+ \Leftrightarrow SO_4^{2-} + H_2O + 2H^+ \qquad \text{(AX2-40)}$$

as SO_3^{2-} is much less abundant than HSO_3^-.

Major pathways for the aqueous phase oxidation of S(IV) to S(VI) as a function of pH are shown in Figure AX2-6. For pH up to about 5.3, H_2O_2 is seen to be the dominant oxidant; above 5.3, O_3, followed by Fe(III) becomes dominant. Higher pHs are expected to be found mainly in marine aerosols. However, in marine aerosols, the chloride-catalyzed oxidation of S(IV) may be more important (Zhang and Millero, 1991; Hoppel and Caffrey, 2005). Because NH_4^+ is so effective in controlling acidity, it affects the rate of oxidation of S(IV) to S(VI) and the rate of dissolution of SO_2 in particles and cloud drops.

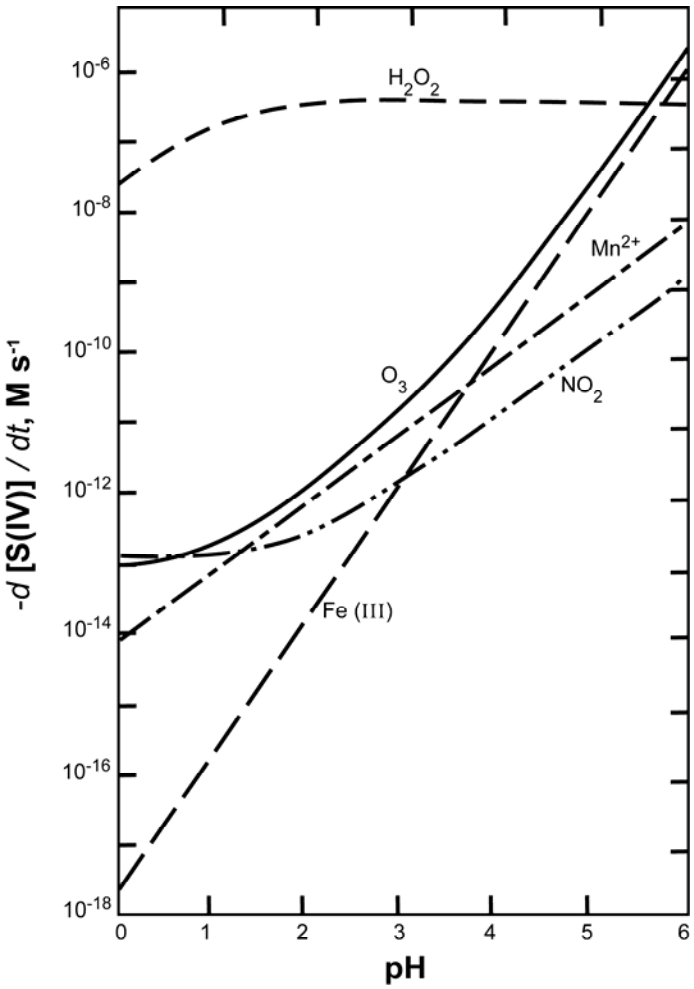


Figure AX2-6. Comparison of aqueous-phase oxidation paths. The rate of conversion of S(IV) to S(VI) is shown as a function of pH. Conditions assumed are: [SO_2(g)] = 5 ppb; [NO_2(g)] = 1 ppb; [H_2O_2(g)] = 1 ppb; [O_3(g)] = 50 ppb; [Fe(III)(aq)] = 0.3 µM; [Mn(II)(aq)] = 0.3 µM.

Source: Seinfeld and Pandis (1998).

Nitrogen dioxide is also taken up in cloud drops and can be oxidized to NO_3^-, although it is much less soluble than SO_2 and this pathway is of minor importance. Instead, the uptake of more highly soluble nitrogen-containing acids initiates aqueous-phase chemistry of NO_3 formation.

Warneck (1999) constructed a box model describing the chemistry of the oxidation of SO_2 and NO_2 including the interactions of N and S species and minor processes in sunlit cumulus clouds. The relative contributions of different reactions to the oxidation of S(IV) species to

S(VI) and NO_2 to NO_3^- 10 minutes after cloud formation are given in Tables AX2-2a and AX2-2b. The two columns show the relative contributions with and without transition metal ions. As can be seen from Table AX2-2a, SO_2 within a cloud (gas + cloud drops) is oxidized mainly by H_2O_2 in the aqueous phase, while and the gas-phase oxidation by OH radicals is small by comparison. A much smaller contribution in the aqueous phase is made by methyl hydroperoxide (CH_3OOH) because it is formed mainly in the gas phase and its Henry's Law constant is several orders of magnitude smaller that of H_2O_2. After H_2O_2, HNO_4 is the major contributor to S(IV) oxidation. The contribution from the gas phase oxidation of SO_2 to be small by comparison to the aqueous -phase reactions given above.

In contrast to the oxidation of SO_2, Table AX2-2b shows that the oxidation of NO_2 occurs mainly in the gas phase within clouds, implying that the gas phase oxidation of NO_2 by OH radicals predominates. Clouds occupy about 15%, on average, of the volume of the troposphere.

The values shown in Tables AX2-2a and AX2-2b indicate that only about 20% of SO_2 is oxidized in the gas phase, but about 90% of NO_2 is oxidized in the gas phase. Thus, SO_2 is oxidized mainly by aqueous-phase reactions, but NO_2 is oxidized mainly by gas phase reactions.

Multiphase Chemical Processes Involving Sulfur Oxides and Ammonia

The phase partitioning of NH_3 with deliquesced aerosol solutions is controlled primarily by the thermodynamic properties of the system expressed as follows:

$$NH_{3g} \overset{K_H}{\leftrightarrow} [NH_{3aq}] \overset{K_b}{\leftrightarrow} [NH_4^+] + K_W/[H^+] \qquad \text{(AX2-41)}$$

where K_H and K_b are the temperature-dependent Henry's Law and dissociation constants (62 M atm^{-1}) (1.8×10^{-5} M), respectively, for NH_3, and K_w is the ion product of water (1.0×10^{-14} M) (Chameides, 1984). It is evident that for a given amount of NH_x (NH_3 + particulate NH_4^+) in the system, increasing aqueous concentrations of particulate H^+ will shift the partitioning of NH_3 towards the condensed phase. Consequently, under the more polluted conditions characterized by higher concentrations of acidic sulfate aerosol, ratios of gaseous NH_3 to particulate NH_4^+ decrease (Smith et al., 2007). It also follows that in marine air, where aerosol acidity varies substantially as a function of particle size, NH_3 partitions preferentially to the more acidic sub-µm size fractions (e.g., Keene et al., 2004; Smith et al., 2007).

Because the dry-deposition velocity of gaseous NH_3 to the surface is substantially greater than that for the sub-μm, sulfate aerosol size factions with which most particulate NH_4^+ is associated, dry-deposition fluxes of total NH_3 are dominated by the gas phase fraction (Russell et al., 2003; Smith et al., 2007). Consequently, partitioning with highly acidic sulfate aerosols effectively increases the atmospheric lifetime of total NH_3 against dry deposition. This shift has important consequences for NH_3 cycling and potential ecological effects. In coastal New England during summer, air transported from rural eastern Canada contains relatively low concentrations of particulate non-sea salt (nss) SO_4^{2-} and total NH_3 (Smith et al., 2007). Under these conditions, the roughly equal partitioning of total NH_3 between the gas and particulate phases sustains substantial dry-deposition fluxes of total NH_3 to the coastal ocean (median of 10.7 μmol m^{-2} day^{-1}). In contrast, heavily polluted air transported from the industrialized midwestern United States contains concentrations of nss SO_4^{2-} and total NH_3 that are, about a factory of 3 greater, based on median values. Under these conditions, most total NH_3 (>85%) partitions to the highly acidic sulfate aerosol size fractions and, consequently, the median dry-deposition flux of total NH_3 is 30% lower than that under the cleaner northerly flow regime. The relatively longer atmospheric lifetime of total NH_3 against dry deposition under more polluted conditions implies that, on average, total NH_3 would accumulate to higher atmospheric concentrations under these conditions and also be subject to atmospheric transport over longer distances. Consequently, the importance NH_x of removal via wet deposition would also increase. Because of the inherently sporadic character of precipitation, we might expect by greater heterogeneity in NH_3 deposition fields and any potential responses by sensitive ecosystems downwind of major S-emission regions.

AX2.5 TRANSPORT OF NITROGEN AND SULFUR OXIDES IN THE ATMOSPHERE

Major episodes of high O_3 concentrations in the eastern United Sates and in Europe are associated with slow moving high-pressure systems. High-pressure systems during the warmer seasons are associated with subsidence, resulting in warm, generally cloudless conditions with light winds. The subsidence results in stable conditions near the surface, which inhibit or reduce the vertical mixing of O_3 precursors (NO_x, VOCs, and CO). Photochemical activity is enhanced because of higher temperatures and the availability of sunlight. However, it is becoming

increasingly apparent that transport of O_3 and NO_x and VOC from distant sources can provide significant contributions to local [O_3] even in areas where there is substantial photochemical production. There are a number of transport phenomena occurring either in the upper boundary layer or in the free troposphere which can contribute to high O_3 values at the surface. These phenomena include stratospheric-tropospheric exchange (STE), deep and shallow convection, low-level jets, and the so-called "conveyor belts" that serve to characterize flows around frontal systems.

Convective Transport

Crutzen and Gidel (1983), Gidel (1983), and Chatfield and Crutzen (1984) hypothesized that convective clouds played an important role in rapid atmospheric vertical transport of trace species and first tested simple parameterizations of convective transport in atmospheric chemical models. At nearly the same time, evidence was shown of venting the boundary layer by shallow, fair weather cumulus clouds (e.g., Greenhut et al., 1984; Greenhut, 1986). Field experiments were conducted in 1985 which resulted in verification of the hypothesis that deep convective clouds are instrumental in atmospheric transport of trace constituents (Dickerson et al., 1987). Once pollutants are lofted to the middle and upper troposphere, they typically have a much longer chemical lifetime and with the generally stronger winds at these altitudes, they can be transported large distances from their source regions. Transport of NO_x from the boundary layer to the upper troposphere by convection tends to dilute the higher in the boundary layer concentrations and extend the NO_x lifetime from less than 24 hours to several days. Photochemical reactions occur during this long-range transport. Pickering et al. (1990) demonstrated that venting of boundary layer NO_x by convective clouds (both shallow and deep) causes enhanced O_3 production in the free troposphere. The dilution of NO_x at the surface can often increase O_3 production efficiency. Therefore, convection aids in the transformation of local pollution into a contribution to global atmospheric pollution. Downdrafts within thunderstorms tend to bring air with less NO_x from the middle troposphere into the boundary layer. Lightning produces NO which is directly injected chiefly into the middle and upper troposphere. The total global production of NO by lightning remains uncertain, but is on the order of 10% of the total.

Observations of the Effects of Convective Transport

The first unequivocal observations of deep convective transport of boundary layer pollutants to the upper troposphere were documented by Dickerson et al. (1987). Instrumentation aboard three research aircraft measured CO, O_3, NO, NO_x, NO_y, and hydrocarbons in the vicinity of an active mesoscale convective system near the Oklahoma/Arkansas border during the 1985 PRE-STORM experiment. Anvil penetrations about two hours after maturity found greatly enhanced mixing ratios inside the cloud of all of the aforementioned species compared with outside it. Nitric oxide mixing ratios in the anvil averaged 3 to 4 ppbv, with individual 3-min observations reaching 6 ppbv; boundary layer NO_x was typically 1.5 ppbv or less outside the cloud. Therefore, the anvil observations represent a mixture of boundary layer NO_x and NO_x contributed by lightning. Luke et al. (1992) summarized the air chemistry data from all 18 flights during PRE-STORM by categorizing each case according to synoptic flow patterns. Storms in the maritime tropical flow regime transported large amounts of CO, O_3, and NO_y into the upper troposphere with the midtroposphere remaining relatively clean. During frontal passages a combination of stratiform and convective clouds mixed pollutants more uniformly into the middle and upper levels.

Prather and Jacob (1997) and Jaegle et al. (1997) noted that precursors of HO_x are also transported to the upper troposphere by deep convection, in addition to primary pollutants (e.g., NO_x, CO, VOCs). The HO_x precursors of most importance are water vapor, HCHO, H_2O_2, CH_3OOH, and acetone. The hydroperoxyl radical is critical for oxidizing NO to NO_2 in the O_3 production process as described above.

Over remote marine areas, the effects of deep convection on trace gas distributions differ from those over moderately polluted continental regions. Chemical measurements taken by the NASA ER-2 aircraft during the Stratosphere-Troposphere Exchange Project (STEP) off the northern coast of Australia show the influence of very deep convective events. Between 14.5 and 16.5 km on the February 2-3, 1987 flight, chemical profiles that included pronounced maxima in CO, water vapor, and CCN, and minima of NO_y, and O_3 (Pickering et al., 1993). Trajectory analysis showed that these air parcels likely were transported from convective cells 800-900 km upstream. Very low marine boundary layer mixing ratios of NO_y and O_3 in this remote region were apparently transported upward in the convection. A similar result was noted in CEPEX (Central Equatorial Pacific Experiment; Kley et al., 1996) and in INDOEX (Indian

Ocean Experiment) (deLaat et al., 1999) where a series of ozonesonde ascents showed very low upper tropospheric O_3 following deep convection. It is likely that similar transport of low-ozone tropical marine boundary layer air to the upper troposphere occurs in thunderstorms along the east coast of Florida. Deep convection occurs frequently over the tropical Pacific. Low-ozone and low-NO_x convective outflow likely will descend in the subsidence region of the subtropical eastern Pacific, leading to some of the cleanest air that arrives at the west coast of the United States.

The discussion above relates to the effects of specific convective events. Observations have also been conducted by NASA aircraft in survey mode, in which the regional effects of many convective events can be measured. The SONEX (Subsonic Assessment Ozone and Nitrogen Oxides Experiment) field program in 1997 conducted primarily upper tropospheric measurements over the North Atlantic. The regional effects of convection over North America and the Western Atlantic on upper tropospheric NO_x were pronounced (Crawford et al., 2000; Allen et al., 2000). A discussion of the results of model calculations of convection and its effects can be found in Section AX2.7.

Effects on Photolysis Rates and Wet Scavenging

Thunderstorm clouds are optically very thick, and, therefore, have major effects on radiative fluxes and photolysis rates. Madronich (1987) provided modeling estimates of the effects of clouds of various optical depths on photolysis rates. In the upper portion of a thunderstorm anvil, photolysis is likely to be enhanced by a factor of 2 or more due to multiple reflections off the ice crystals. In the lower portion and beneath the cloud, photolysis is substantially decreased. With enhanced photolysis rates, the NO/NO_2 ratio in the upper troposphere is driven to larger values than under clear-sky conditions. Existing experimental evidence seems to confirm, at least qualitatively these model results (Kelley et al., 1994).

Thunderstorm updraft regions, which contain copious amounts of water, are regions where efficient scavenging of soluble species can occur (Balkanski et al., 1993). Nitrogen dioxide itself is not very soluble and therefore wet scavenging is not a major removal process for it. However, a major NO_x reservoir species, HNO_3 is extremely soluble. Very few direct field measurements of the amounts of specific trace gases that are scavenged in storms are available. Pickering et al. (2001) used a combination of model estimates of soluble species that did not include wet scavenging and observations of these species from the upper tropospheric outflow

region of a major line of convection observed near Fiji. Over 90% of the and in the outflow air appeared to have been removed by the storm. About 50% of CH_3OOH and about 80% of HCHO had been lost.

Convective processes and small-scale turbulence transport pollutants both upward and downward throughout the planetary boundary layer and the free troposphere. Ozone and its precursors (NO_x, CO, and VOCs) can be transported vertically by convection into upper part of the mixed layer on one day, then transported overnight as a layer of elevated mixing ratios, perhaps by a nocturnal low-level jet, and then entrained into a growing convective boundary layer downwind and brought back to the surface.

Because NO and NO_2 are only slightly soluble, they can be transported over longer distances in the gas phase than can more soluble species which can be depleted by deposition to moist surfaces, or taken up more readily on aqueous surfaces of particles. During transport, they can be transformed into reservoir species such as HNO_3, PANs, and N_2O_5. These species can then contribute to local NO_x concentrations in remote areas. For example, it is now well established that PAN decomposition provides a major source of NO_x in the remote troposphere (Staudt et al., 2003). PAN decomposition in subsiding air masses from Asia over the eastern Pacific could make an important contribution to O_3 and NO_x enhancement in the U.S. (Kotchenruther et al., 2001; Hudman et al., 2004). Further details about mechanisms for transporting ozone and its precursors were described at length in CD06.

AX2.6 SOURCES AND EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE

All three of the species listed in the title to this section have both natural and anthropogenic sources. In Section AX2.6.1, interactions of NO_x with the terrestrial biosphere are discussed. Because of the tight coupling between processes linking emissions and deposition, they are discussed together. In Section AX2.6.2, emissions of NO_x, NH_3, and SO_2 are discussed. Field studies evaluating emissions inventories are discussed in Section AX2.6.3.

AX2.6.1 Interactions of Nitrogen Oxides with the Biosphere

Nitrogen oxides affect vegetated ecosystems, and in turn the atmospheric chemistry of NO_x is influenced by vegetation. Extensive research on nitrogen inputs from the atmosphere to

forests was conducted in the 1980s as part of the Integrated Forest Study, and is summarized by Johnson and Lindberg (1992). The following sections discuss sources of NO_x from soil, deposition of NO_x to foliage, reactions with biogenic hydrocarbons, and ecological effects of nitrogen deposition.

NO_x Sources

Soil NO

Nitric oxide NO from soil metabolism is the dominant, but not exclusive, source of nitrogen oxides from the biosphere to the atmosphere. As noted below, our understanding of NO_2 exchange with vegetation suggests that there should be emission of NO_2 from foliage when ambient concentrations are less than about 1 ppb. However, Lerdau et al. (2000) have pointed out that present understanding of the global distribution of NO_x is not consistent with a large source that would be expected in remote forests if NO_2 emission was important when atmospheric concentrations were below the compensation point.

The pathways for nitrification and denitrification include two gas-phase intermediates, NO and N_2O, some of which can escape. While N_2O is of interest for its greenhouse gas potential and role in stratospheric chemistry it is not considered among the reactive nitrogen oxides important for urban and regional air quality and will not be discussed further. Temperature and soil moisture are critical factors that control the rates of reaction and importantly the partitioning between NO and N_2O which depend on oxygen levels: in flooded soils where oxygen levels are low, N_2O is the dominant soil nitrogen gas; as soil dries, allowing more O_2 to diffuse, NO emissions increase. In very dry soils microbial activity is inhibited and emissions of both N_2O and NO decrease. Nitrogen metabolism in soil is strongly dependent on the substrate concentrations. Where nitrogen is limiting, nitrogen is efficiently retained and little gaseous nitrogen is released. Where nitrogen is in excess of demand, gaseous nitrogen emissions increase; consequently, soil NO emissions are highest in fertilized agriculture and tropical soils (Davidson and Kingerlee, 1997; Williams et al., 1992).

Sinks

Several reactive nitrogen are species are deposited to vegetation, among them, HNO_3, NO_2, PAN, and organic nitrates.

HNO_3

Deposition of HNO_3 appears to be relatively simple. Field observations based on concentration gradients and recently using eddy covariance demonstrate rapid deposition that approaches the aerodynamic limit (as constrained by atmospheric turbulence) in the Wesely (1989) formulation based on analogy to resistance. Surface resistance for HNO_3 uptake by vegetation is negligible. Deposition rates are independent of leaf area or stomatal conductance, implying that deposition occurs to branches, soil, and leaf cuticle as well as internal leaf surfaces.

Deposition velocities (V_d) typically exceed 1 cm $s^{-1}$and exhibit a daily pattern controlled by turbulence characteristics: midday maximum and lower values at night when there is stable boundary layer.

Deposition of NO_2

Nitrogen dioxide interaction with vegetation is more complex. Application of ^{15}N-labeled Nitrogen Dioxide demonstrates that Nitrogen Dioxide is absorbed and metabolized by foliage (Siegwolf et al., 2001; Mocker et al., 1998; Segschneider et al., 1995; Weber, et al., 1995). Exposure to NO_2 induces nitrate reductase (Weber et al., 1995, 1998), a necessary enzyme for assimilating oxidized nitrogen. Understanding of NO_2 interactions with foliage is largely based on leaf cuvette and growth chamber studies, which expose foliage or whole plants to controlled levels of NO_2 and measure the fraction of NO_2 removed from the chamber air. A key finding is that the fit of NO_2 flux to NO_2 concentration, has a non-zero intercept, implying a compensation point or internal concentration. In studies at very low NO_2 concentrations emission from foliage is observed (Teklemariam and Sparks, 2006). Evidence for a compensation point is not solely based on the fitted intercept. Nitrogen dioxide uptake rate to foliage is clearly related to stomatal conductance. Internal resistance is variable, and may be associated with concentrations of reactive species such as ascorbate in the plant tissue that react with NO_2 (Teklemariam and Sparks, 2006). Foliar NO_2 emissions show some dependence on nitrogen content (Teklemariam and Sparks, 2006). Internal NO_2 appears to derive from plant nitrogen metabolism.

Two approaches to modeling NO_2 uptake by vegetation are the resistance-in-series analogy which considers flux (F) as the product of concentration (C) and V_d, where is related to the sum of aerodynamic, boundary layer, and internal resistances (R_a, R_b, and R_c ; positive fluxes are from atmosphere to foliage)

$$F = CV_d \quad \text{(AX2-42)}$$

$$V_d = (R_a + R_b + R_c)^{-1} \quad \text{(AX2-43)}$$

R_a and R_b and controlled by turbulence in the mixed layer; R_c is dependent on characteristics of the foliage and other elements of the soil, and may be viewed as 2 combination of resistance internal to the foliage and external on the cuticle, soils, and bark. This approach is amenable to predicting deposition in regional air quality models (Wesely, 1989). Typically, the NO_2, V_d is less than that for O_3, due to the surface's generally higher resistance to NO_2 uptake, consistent with NO_2's lower reactivity.

Alternatively, NO_2 exchange with foliage can be modeled from a physiological viewpoint where the flux from the leaf is related to the stomatal conductance and a concentration gradient between the ambient air and interstitial air in the leaf. This approach best describes results for exchange with individual foliage elements, and is expressed per unit leaf (needle) area. While this approach provides linkage to leaf physiology, it is not straightforward to scale up from the leaf to ecosystem scale:

$$J = g_s(C_a - C_i) \quad \text{(AX2-44)}$$

This model implicitly associates the compensation point with a finite internal concentration. Typically observed compensation points are around 1 ppb. Finite values of internal NO_2 concentration are consistent with metabolic pathways that include oxides of nitrogen. In this formulation, the uptake will be linear with NO_2 concentration, which is typically observed with foliar chamber studies.

Several studies have shown the UV dependence of NO_2 emission, which implies some photo-induced surface reactions that release NO_2. Rather than model this as a UV-dependent internal concentration, it would be more realistic to add an additional term to account for emission that is dependent on light levels and other surface characteristics:

$$J = g_s(C_a - C_i) = J_s(UV) \quad \text{(AX2-45)}$$

The mechanisms for surface emission are discussed below. Measurement of NO_2 flux is confounded by the rapid interconversion of NO, NO_2, and O_3 (Gao et al., 1991).

PAN Deposition

Peroxyacetyl nitrate is phytotoxic, so clearly it is absorbed at the leaf. Observations based on inference from concentration gradients and rates of decline at night (Shepson et al., 1992; Schrmipf et al., 1996) and leaf chamber studies (Teklemariam and Sparks, 2004) have indicated that PAN uptake is slower than that of O_3; however, recent work in coniferous canopy with direct eddy covariance PAN flux measurements indicated a V_d more similar to that of O_3. Uptake of PAN is under stomatal control, has a non-zero deposition at night, and is influenced by leaf wetness (Turnipseed et al., 2006). On the other hand, flux measurements determined by gradient methods over a grass surface showed a V_d closer to 0.1 cm s^{-1}, with large uncertainty (Doskey et al., 2004). A factor of 10 uncertainty remains in V_d 0.1-1 cm s^{-1} giving a range. Whether the discrepancies are methodological or indicate intrinsic differences between different vegetation is unknown. Uptake of PAN is smaller than its thermal decomposition in all cases.

Organic Nitrates

The biosphere also interacts with NO_x through hydrocarbon emissions and their subsequent reactions to form multi-functional organic nitrates. Isoprene nitrates are an important class of these. Isoprene reacts with OH to form a radical that adds NO_2 to form a hydroxyalkyl nitrate. The combination of hydroxyl and nitrate functional group makes these compounds especially soluble with low vapor pressures; they likely deposit rapidly (Shepson et al., 1996; Treves et al., 2000). Many other unsaturated hydrocarbons react by analogous routes. Observations at Harvard Forest show a substantial fraction of total NO_y not accounted for by NO, NO_2 and PAN, which is attributed to the organic nitrates (Horii et al., 2006, Munger et al., 1998). Furthermore, the total NO_y flux exceeds the sum of HNO_3, NO_x, and PAN, which implies that the organic nitrates are a substantial fraction of nitrogen deposition. Other observations that show evidence of hydoxyalkyl nitrates include those of Grossenbacher et al. (2001) and Day et al. (2003).

Formation of the hydroxyalkyl nitrates occurs after VOC + OH reaction. In some sense, this mechanism is just an alternate pathway for OH to react with NO_x to form a rapidly depositing species. If VOC were not present, OH would be available to react with NO_2 when it is present instead to form HNO_3.

HONO

Nitrous acid formation on vegetative surfaces at night has long been observed based on measurements of positive gradients (Harrison and Kitto, 1994). Surface reactions of NO_2 enhanced by moisture were proposed to explain these results. Production was evident at sites with high ambient NO_2; at low concentration, uptake of HONO exceeded the source. Daytime observations of HONO when rapid photolysis is expected to deplete ambient concentrations to very low levels implies a substantial source of photo-induced HONO formation at a variety of forested sites where measurements have been made. Estimated source strengths are 200-1800 pptv hr^{-1} in the surface layer (Zhou et al., 2002a, 2003), which is about 20 times faster than all nighttime sources. Nitrous acid sources could be important to OH/HO_2 budgets as HONO is rapidly photolyzed by sunlight to OH and NO. Additional evidence of light-dependent reactions to produce HONO comes from discovery of a HONO artifact in pyrex sample inlet lines exposed to ambient light. Either covering the inlet or washing it eliminated the HONO formation (Zhou et al., 2002b). Similar reactions might serve to explain observations of UV-dependent production of NO_x in empty foliar cuvettes that had been exposed to ambient air (Hari et al., 2003; Raivonen et al., 2003).

Production of HONO in the dark is currently believed to occur via a heterogeneous reaction involving NO_2 on wet surfaces (Jenkin et al., 1988; Pitts et al., 1984; He et al., 2006; Sakamaki et al., 1983), and it is proposed that the mechanism has first-order dependence in both NO_2 and H_2O (Kleffmann et al., 1998; Svensson et al., 1987) despite the stoichiometry. However, the molecular pathway of the mechanism is still under debate. Jenkin et al. (1988) postulated a $H_2O{\cdot}NO_2$ water complex reacting with gas phase NO_2 to produce HONO, which is inconsistent with the formation of an N_2O_4 intermediate leading to HONO as proposed by Finlayson-Pitts et al. (2003). Another uncertainty is whether the reaction forming HONO is dependent on water vapor (Svensson et al., 1987; Stutz et al., 2004) or water adsorbed on surfaces (Kleffmann et al., 1998). Furthermore, the composition of the surface and the available amount of surface or surface-to-volume ratio can significantly influence the HONO production rates (Kaiser and Wu, 1977; Kleffmann et al., 1998; Svensson et al., 1987), which may explain the difference in the rates observed between laboratory and atmospheric measurements.

There is no consensus on a chemical mechanism for photo-induced HONO production. Photolysis of HNO_3 or NO_3^- absorbed on ice or in surface water films has been proposed

(Honrath et al., 2002; Ramazan et al., 2004; Zhou et al., 2001, 2003). Alternative pathways include NO_2 interaction with organic surfaces such as humic substances (George et al., 2005; Stemmler et al., 2006). Note that either NO_3^- photolysis or heterogeneous reaction of NO_2 are routes for recycling deposited nitrogen oxides back to the atmosphere in an active form. Nitrate photolysis would return nitrogen that heretofore was considered irreversibly deposited, surface reactions between NO_2 and water films or organic molecules would decrease the effectiveness of observed NO_2 deposition if the HONO were re-emitted.

Fast Homogeneous Reactions

Inferences from observations at Blodgett Forest (Cohen et al. in prep) suggest that radicals from O_3 + VOC react with NO_x in the canopy to produce HNO_3 and organic nitrates among other species. This mechanism would contribute to canopy retention of soil NO emission in forests with high VOC possibly more effectively than the NO to NO_2 conversion and foliar uptake of NO_2 that has been proposed to reduce the amount of soil NO that escapes to the supra-canopy atmosphere (Jacob and Bakwin, 1991).

Some NO_2 and HNO_3 Flux Data from Harvard Forest

Observations from TDL Measurements of NO_2

Harvard Forest is a rural site in central Massachusetts, where ambient NO_x, NO_y, and other pollutant concentrations and fluxes of total NO_y have been measured since 1990 (Munger et al., 1996). An intensive study in 2000 utilized a Tunable Diode Laser Absorption Spectrometer (TDLAS) to measure NO_2 and HNO_3. TDLAS has an inherently fast response, and for species such as NO_2 and HNO_3 with well-characterized spectra it provides an absolute and specific measurement. Absolute concentrations of HNO_3 were measured, and the flux inferred based on the dry deposition inferential method that uses momentum flux measurements to compute a deposition velocity and derives an inferred flux (Wesely and Hicks, 1977; Hicks et al., 1987). Direct eddy covariance calculations for HNO_3 were not possible because the atmospheric variations were attenuated by interaction with the inlet walls despite very short residence time and use of fluorinated silane coatings to make the inlet walls more hydrophobic. Nitrogen Oxide response was adequate to allow both concentration and eddy covariance flux determination. Simultaneously, NO and NO_y eddy covariance fluxes were determined with two separate O_3

chemiluminescence detectors, one equipped with a H_2-gold catalyst at the inlet to convert all reactive nitrogen compounds to NO. Additionally, the measurements include concentration gradients for NO, NO_2, and O_3 over several annual cycles to examine their vertical profiles in the forest canopy.

Overall, the results show typical NO_2 concentrations of 1 ppb under clean-air conditions and mean concentrations up to 3 ppb at night and 1 ppb during daytime for polluted conditions. Net positive fluxes (emission) of NO_2 were evident in the daytime and negative fluxes (deposition) were observed at night (Figure AX2-7). Nitric oxide fluxes were negative during the daytime and near zero at night.

In part the opposite NO and NO_2 fluxes are simply consequences of variable NO/NO_2 distributions responding to vertical gradients in light intensity and O_3 concentration, which resulted in no net flux of NO_x (Gao et al., 1993). In the Harvard Forest situation, the NO and NO_2 measurements were not at the same height above the canopy, and the resulting differences derive at least in part from the gradient in flux magnitude between the two inlets (Figure AX2-8).

At night, when NO concentrations are near 0 due to titration by ambient O_3 there is not a flux of NO to offset NO_2 fluxes. Nighttime data consistently show NO_2 deposition (Figure AX2-9), which increases with increasing NO_2 concentrations. Concentrations above 10 ppb were rare at this site, but the few high NO_2 observations suggest a nonlinear dependence on concentration. The data fit a model with V_d of −0.08 plus an enhancement term that was second order in NO_2 concentration. The second order term implies that NO_2 deposition rates to vegetation in polluted urban sites would be considerably larger than what was observed at this rural site.

After accounting for the NO-NO_2 null cycle the net NO_x flux could be derived. Overall, there was a net deposition of NO_x during the night and essentially zero flux in the day, with large variability in the magnitude and sign of individual flux observations (Figure AX2-10). For the periods with [NO_2] > 2 ppb, deposition was always observed. These canopy-scale field observations are consistent with a finite compensation point for NO_2 in the canopy that offsets foliar uptake or even reverses it when concentrations are especially low. At concentrations above the compensation point, NO_x is absorbed by the canopy. Examination of concentration profiles corroborates the flux measurements (Figure AX2-11). During daytime for low-NO_x conditions, there is a local maximum in the concentration profile near the top of the canopy

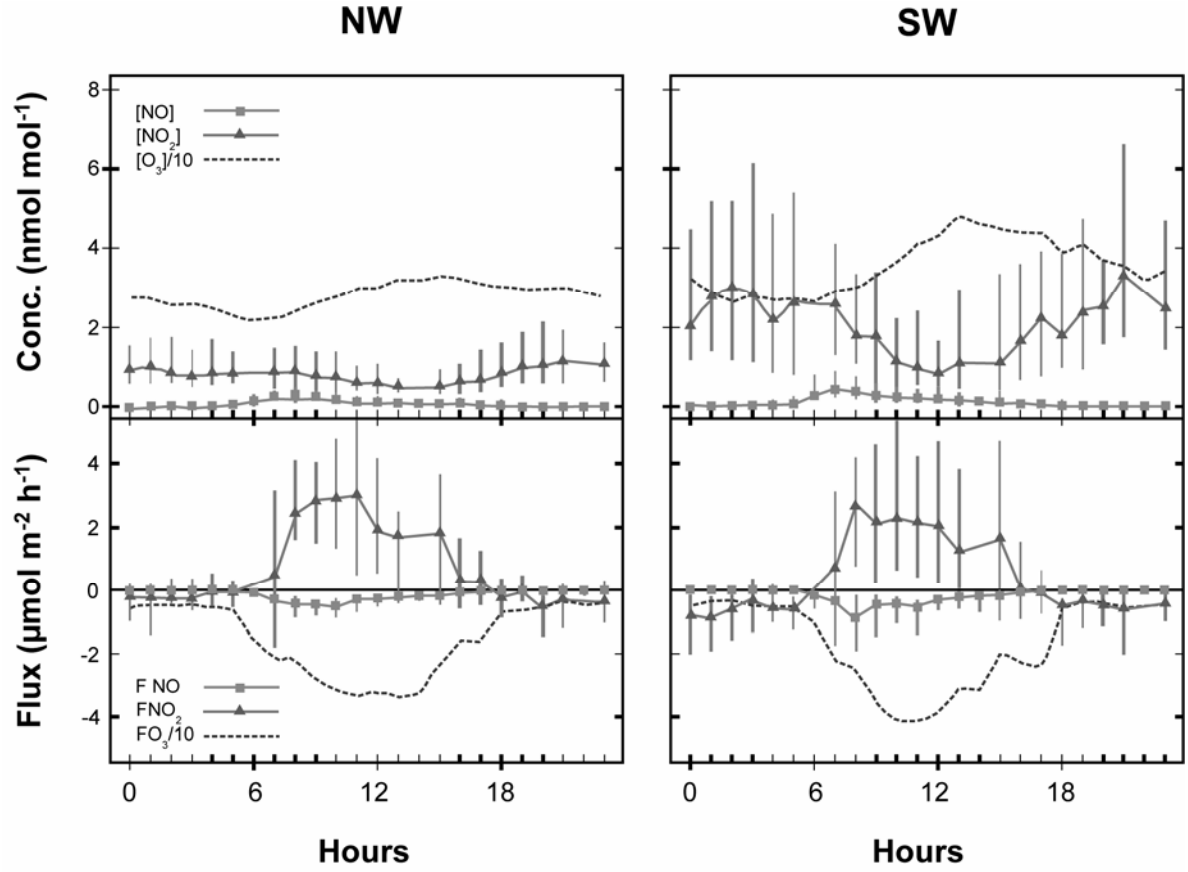


Figure AX2-7. **Diel cycles of median concentrations (upper panels) and fluxes (lower panels) for the Northwest clean sector, left panels) and Southwest (polluted sector, right panels) wind sectors at Harvard Forest, April-November, 2000, for NO, NO_2, and O_3/10. NO and O_3 were sampled at a height of 29 m, and NO_2 at 22 m. Vertical bars indicate 25th and 27th quartiles for NO and NO_2 measurements. NO_2 concentration and nighttime deposition are enhanced under southwesterly conditions, as are O_3 and the morning NO maximum.**

Source: Horii et al. (2004).

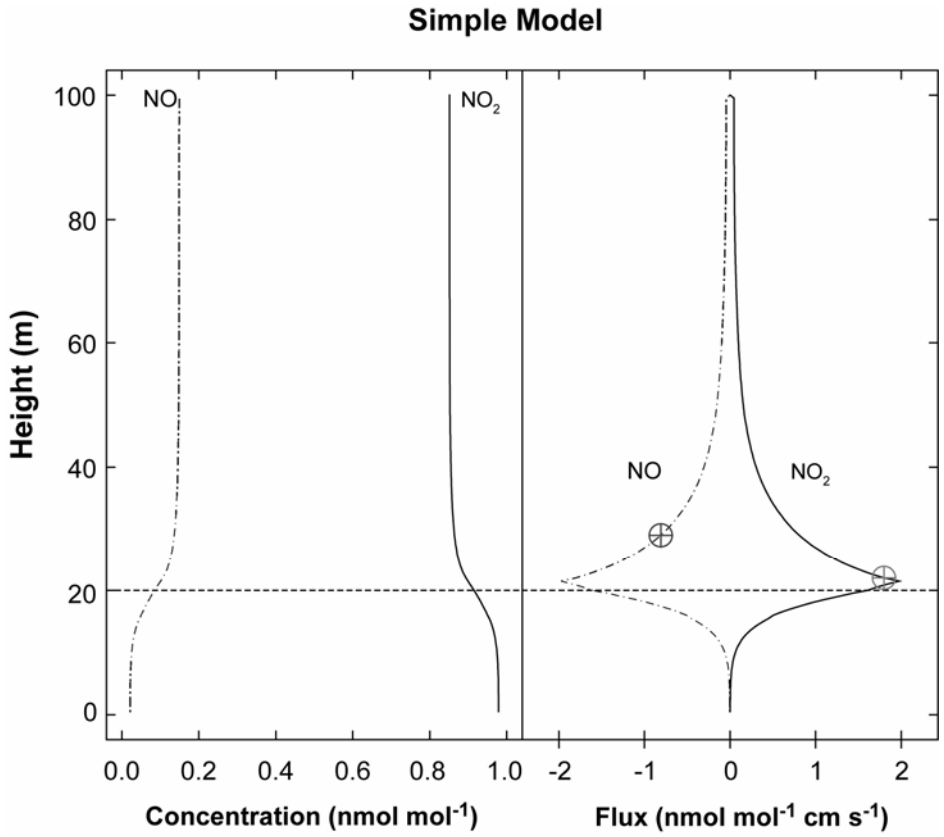


Figure AX2-8. **Simple NO_x photochemical canopy model outputs. Left panel, concentrations of NO (dashed) and NO_2 (solid); right, fluxes of NO (dashed) and NO_2 (solid). Symbols indicate measurement heights for NO (29m) and NO_2 (22m) at Harvard Forest. The model solves the continuity equation for NO concentration at 200 levels, d/dz(-Kc(dNO/dz)) = PNO − LNO, where PNO = [NO]/t1, LNO = [NO]/t2, and zero net deposition or emission of NO_x is allowed. NO_x (NO + NO_2) is normalized to 1ppb. t1 = 70s in this example. Due to the measurement height difference, observed upward NO_2 flux due to photochemical cycling alone should be substantially larger than observed downward NO flux attributable to the same process.**

Source: Horii (2002).

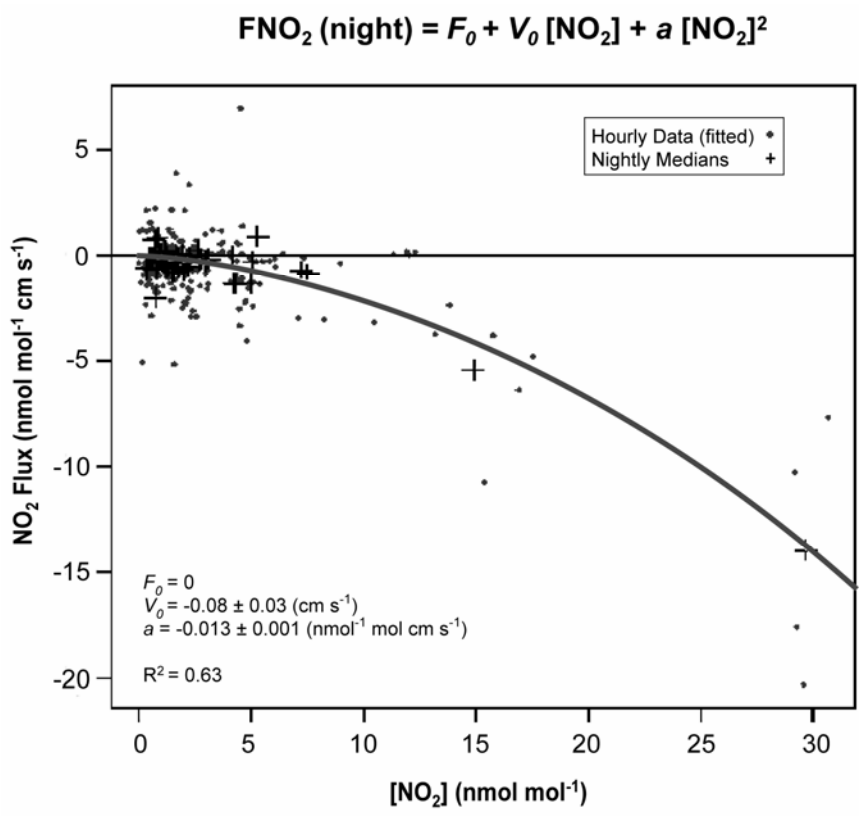


Figure AX2-9. Hourly (dots) and median nightly (pluses) NO_2 flux vs. concentration, with results of least-squares fit on the hourly data (curve). The flux is expressed in units of concentration times velocity (nmol mol^{-1} cm s^{-1}) in order to simplify the interpretation of the coefficients in the least-squares fit. Pressure and temperature corrections have been taken into account in the conversion from density to mixing ratio.

Source: Horii et al. (2004).

where O_3 has a local minimum, which is consistent with foliar emission or light-dependent production of NO_x in the upper canopy. Depletion is evident for both NO_x and O_3 near the forest floor. Air reaching the ground has passed through the canopy where uptake is efficient and the vertical exchange rates near the ground are slow. At night, the profiles generally decrease with decreasing height above the ground, showing only uptake. At higher concentrations, the daytime NO_x concentrations are nearly constant through the canopy; no emission is evident from the sunlit leaves.

Figure AX2-12 compares observed fluxes of all the observed species. The measured NO_x and estimated PAN fluxes are small relative to the observed total NO_y flux. In clean air, HNO_3

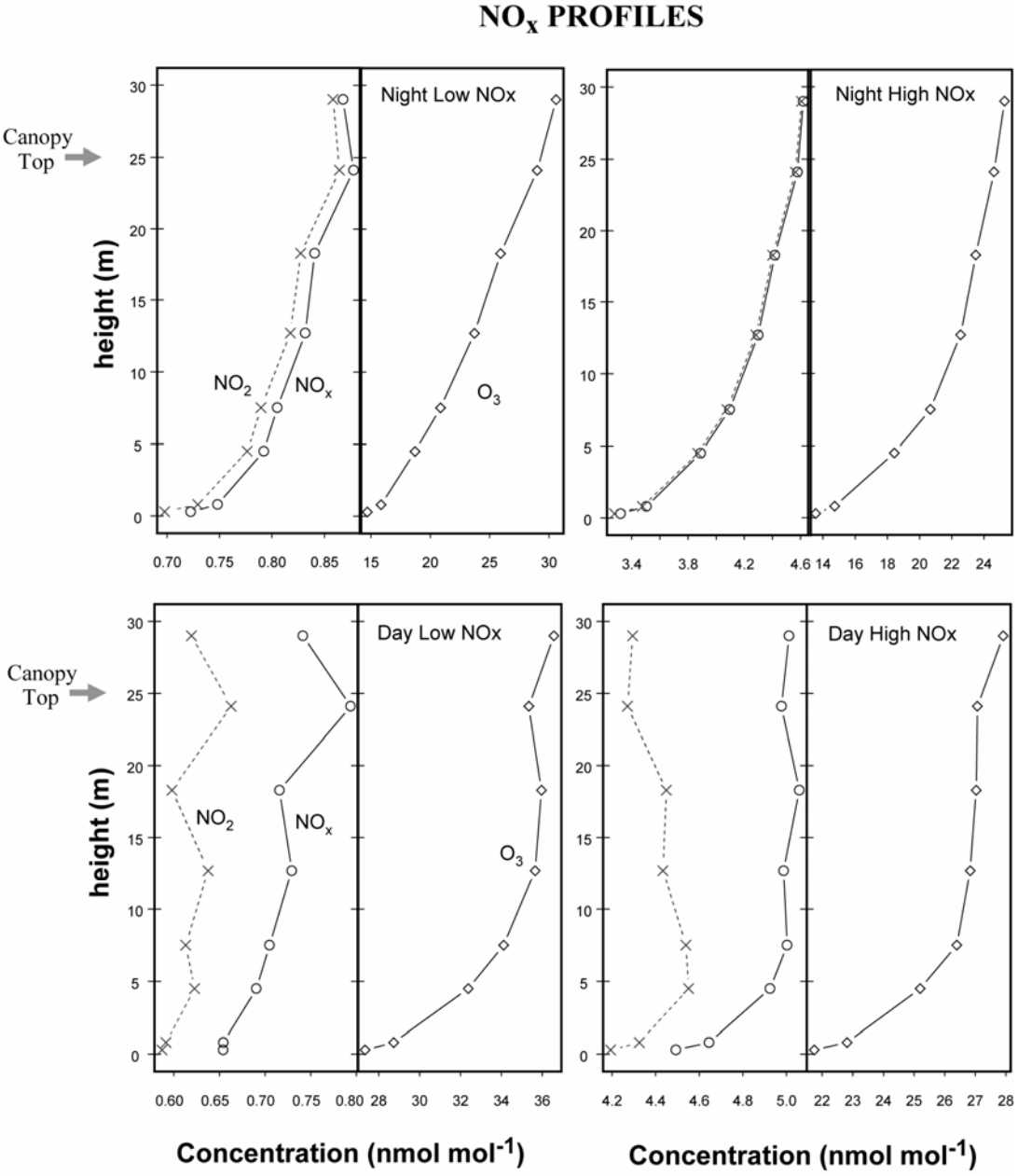


Figure AX2-11. Averaged profiles at Harvard Forest give some evidence of some NO_2 input near the canopy top from light-mediated ambient reactions, or emission from open stomates.

Source: Horii et al. (2004).

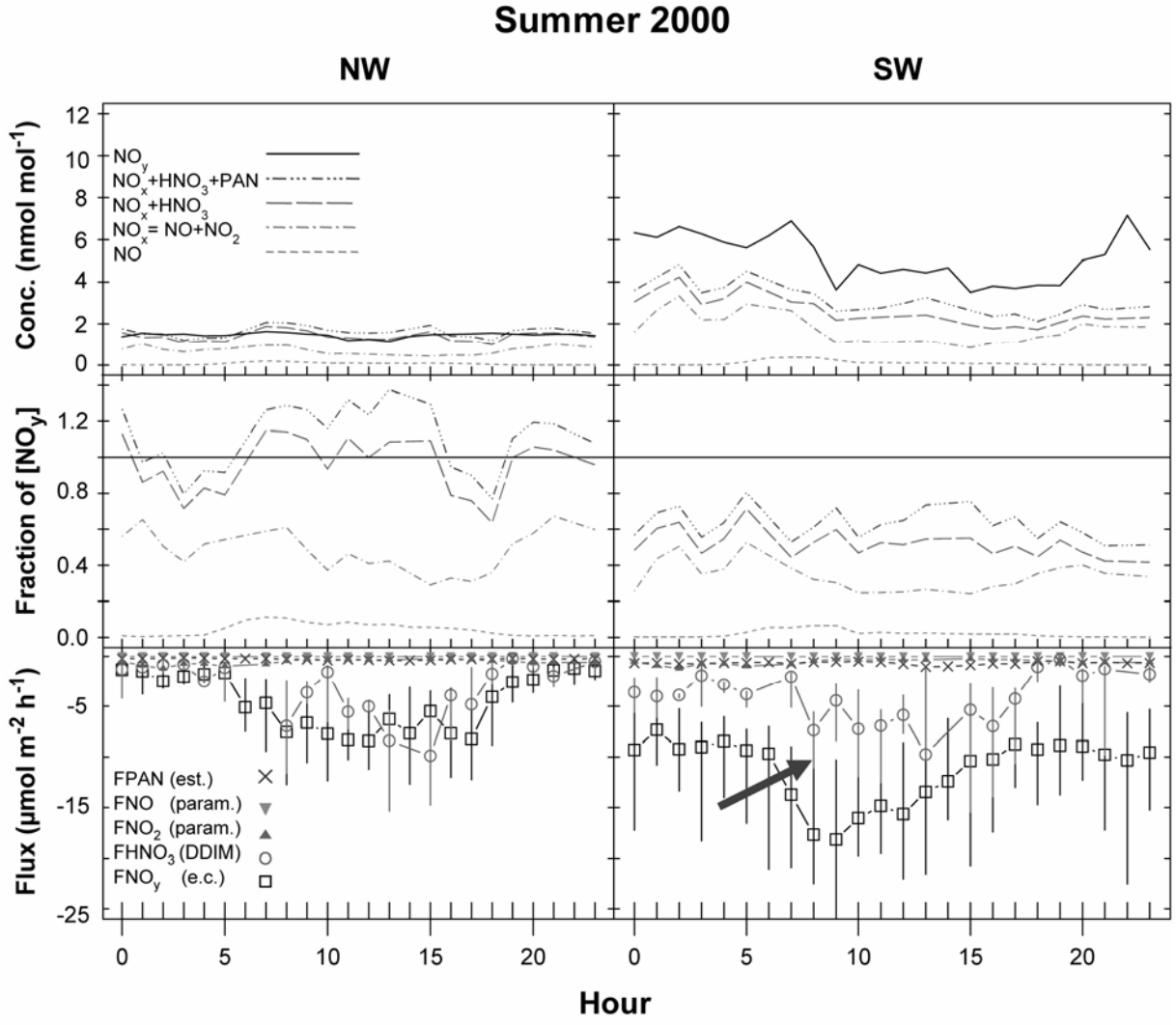


Figure AX2-12. **Summer (June-August) 2000 median concentrations (upper panels), fractions of NO_y (middle panels), and fluxes (lower panels) of NO_y and component species separated by wind direction (Northwest on the left and Southwest on the right). Vertical lines in the flux panels show 25th and 75th quartiles of $F(NO_y)$ and $F(HNO_3)$; negative fluxes represent deposition; $F(NO_x)$ is derived from eddy covariance F(NO) and $F(NO_2)$ measurements (corrected for photochemical cycling), $F(HNO_3)$ is inferred, and $F(NO_y)$ was measured by eddy covariance. The sum of NO_x, HNO_3, and PAN accounts for all of the NO_y concentration and flux for Northwesterly (unpolluted background) flows, whereas up to 50% of NO_y and $F(NO_y)$ under Southwesterly flows are in the form of reactive nitrogen species whose fluxes are not measured or estimated here.**

Source: Horii et al. (2006).

accounts for nearly all the NO_y flux and the sum of all measured species is about equal to the NO_y concentration. However, in polluted conditions, unmeasured species are up to 25% of the NO_y, and HNO_3 fluxes cannot account for all the total NO_y flux observed. Likely these unmeasured NO_y species are hydroxyalkyl nitrates and similar compounds and are rapidly deposited. Although NO_2 uptake may be important to the plant, because it is an input directly to the interior of foliage that can be used immediately in plant metabolism, it is evidently not a significant part of overall nitrogen deposition to rural sites. The deposition of HNO_3 and multifunctional organic nitrates are the largest elements of the nitrogen dry deposition budget. Two key areas of remaining uncertainty are the production of HONO over vegetation and the role of very reactive biogenic VOCs. HONO is important because its photolysis is a source of OH radicals, and its formation may represent an unrecognized mechanism to regenerate photochemically active NO_x from nitrate that had been considered terminally removed from the atmosphere.

Ecosystem Effects

In addition to the contribution to precipitation acidity, atmospheric nitrogen oxides have ecological effects. Total loading by both and wet and dry deposition is the relevant metric for considering ecosystem impacts. At low inputs, nitrogen deposition adds essential nutrients to terrestrial ecosystems. Most temperate forests are nitrogen limited; thus the inputs stimulate growth. Anthropogenic nitrogen may influence some plant species different and alter the distribution of plant species (cf. Wedin and Tilman, 1996). At high nitrogen loading, where nitrogen inputs exceed nutrient requirements, deleterious effects including forest decline associated with 'nitrogen saturation' are seen (Aber at al., 1998; Driscoll et al., 2003). In aquatic ecosystems, however, nitrogen is may or may not be limiting, but in brackish waters atmospheric deposition of anthropogenic nitrogen is suspected of contributing to eutrophication of some coastal waters and lakes (see Bergstrom and Jansson, 2006; Castro and Driscoll, 2002).

AX2.6.2 Emissions of NO_x, NH_3, and SO_2

Emissions of NO_x

Estimated annual emissions of NO_x, NH_3, and SO_2 for 2002 (U.S. Environmental Protection Agency, 2006) are shown in Table AX2-3. Methods for estimating emissions of

criteria pollutants, quality assurance procedures, and examples of emissions calculated by using data are given in U.S. Environmental Protection Agency (1999). Discussions of uncertainties in current emissions inventories and strategies for improving them can be found in NARSTO (2005).

As can be seen from the table, combustion by stationary sources, such as electrical utilities and various industries, accounts for roughly half of total anthropogenic emissions of NO_x. Mobile sources account for the other half, with highway vehicles representing the major mobile source component. Approximately half the mobile source emissions are contributed by diesel engines, the remainder are emitted by gasoline-fueled vehicles and other sources.

Emissions of NO_x associated with combustion arise from contributions from both fuel nitrogen and atmospheric nitrogen. Combustion zone temperatures greater than about 1300 K are required to fix atmospheric N_2:

$$N_2 + O_2 \rightarrow 2NO \qquad \text{(AX2-46)}$$

Otherwise, NO can be formed from fuel N according to this reaction:

$$C_aH_bO_cN_d + O_2 \rightarrow xCO_2 + yH_2O + zNO \qquad \text{(AX2-47)}$$

In addition to NO formation by the schematic reactions given above, some NO_2 and CO are also formed depending on temperatures, concentrations of OH and HO_2 radicals and O_2 levels. Fuel nitrogen is highly variable in fossil fuels, ranging from 0.5 to 2.0 percent by weight (wt %) in coal to 0.05% in light distillates (e.g., diesel fuel), to 1.5 wt % in heavy fuel oils (UK AQEG, 2004). The ratio of NO_2 to NO_x in primary emissions ranges from 3 to 5 % from gasoline engines, 5 to 12% from heavy-duty diesel trucks, 5 to 10% from vehicles fueled by compressed natural gas and from 5 to 10% from stationary sources. In addition to NO_x, motor vehicles also emit HONO, with ratios of HONO to NO_x ranging from 0.3% in the Caldecott Tunnel, San Francisco Bay (Kirchstetter and Harley, 1996) to 0.5 to 1.0% in studies in the United Kingdom (UK AQEG, 2004). The NO_2 to NO_x ratios in emissions from turbine jet engines are as high as 32 to 35 % during taxi and takeoff (CD93). Sawyer et al. (2000) have reviewed the factors associated with NO_x emissions by mobile sources. Marine transport

represents a minor source of NO_x, but it constitutes a larger source in the EU where it is expected to represent about two-thirds of land-based sources (UK AQEG, 2004).

NO_x Emissions from Natural Sources (Soil, Wild Fires, and Lightning)

Soil

Emission rates of NO from cultivated soil depend mainly on fertilization levels and soil temperature. About 60% of the total NO_x emitted by soils occurs in the central corn belt of the United States. The oxidation of NH_3, emitted mainly by livestock and soils, leads to the formation of NO, **also** NH_4^+ and NO_3^- fertilizers lead to NO emissions from soils. Estimates of emissions from natural sources are less certain than those from anthropogenic sources. On a global scale, the contribution of soil emissions to the oxidized nitrogen budget is on the order of 10% (van Aardenne et al., 2001; Finlayson-Pitts and Pitts, 2000; Seinfeld and Pandis, 1998), but NO_x emissions from fertilized fields are highly variable. Soil NO emissions can be estimated from the fraction of the applied fertilizer nitrogen emitted as NO_x, but the flux varies strongly with land use and temperature. Estimated globally averaged fractional applied nitrogen loss as NO varies from 0.3% (Skiba et al., 1997) to 2.5% (Yienger and Levy, 1995). Variability within biomes to which fertilizer is applied, such as shortgrass versus tallgrass prairie, accounts for a factor of three in uncertainty (Williams et al., 1992; Yienger and Levy, 1995; Davidson and Kingerlee, 1997).

The local contribution can be much greater than the global average, particularly in summer and especially where corn is grown extensively. Williams et al. (1992) estimated that contributions to NO budgets from soils in Illinois are about 26% of the emissions from industrial and commercial processes in that State. In Iowa, Kansas, Minnesota, Nebraska, and South Dakota, all states with smaller human populations, soil emissions may dominate the NO budget. Conversion of NH_3 to NO_3 (nitrification) in aerobic soils appears to be the dominant pathway to NO. The mass and chemical form of nitrogen (reduced or oxidized) applied to soils, the vegetative cover, temperature, soil moisture, and agricultural practices such as tillage all influence the amount of fertilizer nitrogen released as NO.

Emissions of NO from soils peak in summer when O_3 formation is also at a maximum. An NRC panel report (NRC, 2002) outlined the role of agriculture in emissions of air pollutants including NO and NH_3. That report recommends immediate implementation of best

management practices to control these emissions, and further research to quantify the magnitude of emissions and the impact of agriculture on air quality. Civerolo and Dickerson (1998) report that use of the no-till cultivation technique on a fertilized cornfield in Maryland reduced NO emissions by a factor of seven.

NO_x from Biomass Burning

During biomass burning, nitrogen is derived mainly from fuel nitrogen and not from atmospheric N_2, since temperatures required to fix atmospheric N_2 are likely to be found only in the flaming crowns of the most intense boreal forest fires. Nitrogen is present mainly in plants as amino (NH_2) groups in amino acids. During combustion, nitrogen is released mainly in unidentified forms, presumably as N_2, with very little remaining in fuel ash. Apart from N_2, the most abundant species in biomass burning plumes is NO. Emissions of NO account for only about 10 to 20% relative to fuel N (Lobert et al., 1991). Other species such as NO_2, nitriles, ammonia, and other nitrogen compounds account for a similar amount. Emissions of NO_x are about 0.2 to 0.3% relative to total biomass burned (e.g., Andreae, 1991; Radke et al., 1991). Westerling et al. (2006) have noted that the frequency and intensity of wildfires in the western U.S. have increased substantially since 1970.

Lightning Production of NO

Annual global production of NO by lightning is the most uncertain source of reactive nitrogen. In the last decade, literature values of the global average production rate range from 2 to 20 Tg N per year. However, the most likely range is from 3 to 8 Tg N per year, because the majority of the recent estimates fall in this range. The large uncertainty stems from several factors: (1) a large range of NO production rates per meter of flash length (as much as two orders of magnitude); (2) the open question of whether cloud-to-ground (CG) flashes and intracloud flashes (IC) produce substantially different amounts of NO; (3) the global flash rate; and (4) the ratio of the number of IC flashes to the number of CG flashes. Estimates of the amount of NO produced per flash have been made based on theoretical considerations (e.g., Price et al., 1997), laboratory experiments (e.g., Wang et al., 1998); field experiments (e.g., Stith et al., 1999; Huntrieser et al., 2002, 2007) and through a combination of cloud-resolving model simulations, observed lightning flash rates, and anvil measurements of NO (e.g., DeCaria et al., 2000, 2005; Ott et al., 2007). The latter method was also used by Pickering et al. (1998), who showed that

only ~5 to 20% of the total NO produced by lightning in a given storms exists in the boundary layer at the end of a thunderstorm. Therefore, the direct contribution to boundary layer O_3 production by lightning NO is thought to be small. However, lightning NO production can contribute substantially to O_3 production in the middle and upper troposphere. DeCaria et al. (2005) estimated that up to 10 ppbv of ozone was produced in the upper troposphere in the first 24 hours following a Colorado thunderstorm due to the injection of lightning NO. A series of midlatitude and subtropical thunderstorm events have been simulated with the model of DeCaria et al. (2005), and the derived NO production per CG flash averaged 500 moles/flash while average production per IC flash was 425 moles/flash (Ott et al., 2006).

A major uncertainty in mesoscale and global chemical transport models is the parameterization of lightning flash rates. Model variables such as cloud top height, convective precipitation rate, and upward cloud mass flux have been used to estimate flash rates. Allen and Pickering (2002) have evaluated these methods against observed flash rates from satellite, and examined the effects on ozone production using each method.

Uses of Satellite Data to Derive Emissions

Satellite data have been shown to be useful for optimizing estimates of emissions of NO_2. (Leue et al., 2001; Martin et al., 2003; Jaegle et al., 2005). Satellite-borne instruments such as GOME (Global Ozone Monitoring Experiment; Martin et al., 2003; and references therein) and SCIAMACHY (Scanning Imaging Absorption Spectrometer for Atmospheric Chartography; Bovensmann et al., 1999) retrieve tropospheric columns of NO_2, which can then be combined with model-derived chemical lifetimes of NO_x to yield emissions of NO_x.

Top-down inference of NO_x emission inventory from the satellite observations of NO_2 columns by mass balance requires at minimum three pieces of information: the retrieved tropospheric NO_2 column, the ratio of tropospheric NO_x to NO_2 columns, and the NO_x lifetime against loss to stable reservoirs. A photochemical model has been used to provide information on the latter two pieces of information. The method is generally applied exclusively to land surface emissions, excluding lightning. Tropospheric NO_2 columns are insensitive to lightning NO_x emissions since most of the lightning NO_x in the upper troposphere is present as NO at the local time of the satellite measurements (Ridley et al., 1996), owing to the slower reactions of NO with O_3 there.

Jaeglé et al. (2005) applied additional information on the spatial distribution of emissions and on fire activity to partition NO_x emissions into sources from fossil fuel combustion, soils, and biomass burning. Global a posteriori estimates of soil NO_x emissions are 68% larger than the a priori estimates. Large increases are found for the agricultural region of the western United States during summer, increasing total U.S. soil NO_x emissions by a factor of 2 to 0.9 Tg N yr^{-1}. Bertram et al. (2005) found clear signals in the SCIAMACHY observations of short intense NO_x pulses following springtime fertilizer application and subsequent precipitation over agricultural regions of the western United States. For the agricultural region in North-Central Montana, they calculate a yearly SCIAMACHY top-down estimate that is 60% higher than a commonly used model of soil NO_x emissions by Yienger and Levy (1995).

Martin et al. (2006) retrieved tropospheric nitrogen dioxide (NO_2) columns for May 2004 to April 2005 from the SCIAMACHY satellite instrument to derive top-down NO_x emissions estimates via inverse modeling with a global chemical transport model (GEOS-Chem). The top-down emissions were combined with a priori information from a bottom-up emission inventory with error weighting to achieve an improved a posteriori estimate of the global distribution of surface NO_x emissions. Their a posteriori inventory improves the GEOS-Chem simulation of NO_x, PAN, and HNO_3 with respect to airborne in situ measurements over and downwind of New York City. Their a posteriori inventory shows lower NO_x emissions from the Ohio River valley during summer than during winter, reflecting recent controls on NO_x emissions from electric utilities. Their a posteriori inventory is highly consistent ($R^2 = 0.82$, bias = 3%) with the NEI99 inventory for the United States. In contrast, their a posteriori inventory is 68% larger than a recent inventory by Streets et al. (2003) for East Asia for the year 2000.

Emissions of NH_3

Emissions of NH_3 show a strikingly different pattern from those of NO_x. Three-way catalysts used in motor vehicles emit small amounts of NH_3 as a byproduct during the reduction of NO_x. Stationary combustion sources make only a small contribution to emissions of NH_3 because efficient combustion favors formation of NO_x and, NH_3 from combustion is produced mainly by inefficient, low temperature fuel combustion. For these reasons, most emissions of NH_3 arise from fertilized soils and from livestock.

The initial step in the oxidation of atmospheric NH_3 to NO is by reaction with OH radicals. However, the lifetime of NH_3 from this pathway is sufficiently long (~1-2 months using typical OH values 1-2 × 10^6/cm^3) that it is a small sink compared to uptake of NH_3 by cloud drops, dry deposition, and aerosol particles. Thus, the gas-phase oxidation of NH_3 makes a very small contribution as a source of NO. Holland et al. (2005) estimated wet and dry deposition of NH_x, based on measurements over the continental U.S., and found that emissions of NH_3 in the National Emissions Inventory are perhaps underestimated by about a factor of two to three. Reasons for this imbalance include under-representation of deposition monitoring sites in populated areas and the neglect of off-shore transport in their estimate. The use of fixed deposition velocities that do not reflect local conditions at the time of measurement introduces additional uncertainty into their estimates of dry deposition.

Emissions of SO_2

As can be seen from Table AX2-3, emissions of SO_2 are due mainly to the combustion of fossil fuels by electrical utilities and industry. Transportation related sources make only a minor contribution. As a result, most SO_2 emissions originate from point sources. Since sulfur is a volatile component of fuels, it is almost quantitatively released during combustion and emissions can be calculated on the basis of the sulfur content of fuels to greater accuracy than for other pollutants such as NO_x or primary PM.

The major natural sources of SO_2 are volcanoes and biomass burning and DMS oxidation over the oceans. SO_2 constitutes a relatively minor fraction (0.005% by volume) of volcanic emissions (Holland, 1978). The ratio of H_2S to SO_2 is highly variable in volcanic gases. It is typically much less than one, as in the Mt. Saint Helen's eruption (Turco et al., 1983). However, in addition to being degassed from magma, H_2S can be produced if ground waters, especially those containing organic matter, come into contact with volcanic gases. In this case, the ratio of H_2S to SO_2 can be greater than one. H_2S produced this way would more likely be emitted through side vents than through eruption columns (Pinto et al., 1989). Primary particulate sulfate is a component of marine aerosol and is also produced by wind erosion of surface soils.

Volcanic sources of SO_2 are limited to the Pacific Northwest, Alaska, and Hawaii. Since 1980, the Mount St. Helens volcano in the Washington Cascade Range (46.20 N, 122.18 W, summit 2549 m asl) has been a variable source of SO_2. Its major effects came in the explosive eruptions of 1980, which primarily affected the northern part of the mountainous western half of

the US. The Augustine volcano near the mouth of the Cook Inlet in southwestern Alaska (59.363 N, 153.43 W, summit 1252 m asl) has had variable SO_2 emission since its last major eruptions in 1986. Volcanoes in the Kamchatka peninsula of eastern region of Siberian Russia do not significantly effect surface SO_2 concentrations in northwestern North America. The most serious effects in the U.S. from volcanic SO_2 occurs on the island of Hawaii. Nearly continuous venting of SO_2 from Mauna Loa and Kilauea produces SO_2 in such large amounts that >100 km downwind of the island SO_2 concentrations can exceed 30 ppbv (Thornton and Bandy, 1993). Depending on wind direction, the west coast of Hawaii (Kona region) has had significant deleterious effects from SO_2 and acidic sulfate aerosols for the past decade.

Emissions of SO_2 from burning vegetation are generally in the range of 1 to 2% of the biomass burned (see e.g., Levine et al., 1999). Sulfur is bound in amino acids in vegetation. This organically bound sulfur is released during combustion. However, unlike nitrogen, about half of the sulfur initially present in vegetation is found in the ash (Delmas, 1982). Gaseous emissions are mainly in the form of SO_2 with much smaller amounts of H_2S and OCS. The ratio of gaseous nitrogen to sulfur emissions is about 14, very close to their ratio in plant tissue (Andreae, 1991). The ratio of reduced nitrogen and sulfur species such as NH_3 and H_2S to their more oxidized forms, such as NO and SO_2, increases from flaming to smoldering phases of combustion, as emissions of reduced species are favored by lower temperatures and O_2 reduced availability.

Emissions of reduced sulfur species are associated typically with marine organisms living either in pelagic or coastal zones and with anaerobic bacteria in marshes and estuaries. Mechanisms for their oxidation were discussed in Section AX2.2. Emissions of dimethyl sulfide (DMS) from marine plankton represent the largest single source of reduced sulfur species to the atmosphere (e.g., Berresheim et al., 1995). Other sources such as wetlands and terrestrial plants and soils probably account for less than 5% of the DMS global flux, with most of this coming from wetlands.

The coastal and wetland sources of DMS have a dormant period in the fall/winter from senescence of plant growth. Marshes die back in fall and winter, so dimethyl sulfide emissions from them are lower, reduced light levels in winter at mid to high latitudes reduce cut phytoplankton growth which also tends to reduce DMS emissions. Western coasts at mid to high latitudes have reduced levels of the light that drive photochemical production and oxidation of

DMS. Freezing at mid and high latitudes affects the release of biogenic sulfur gases, particularly in the nutrient-rich regions around Alaska. Transport of SO_2 from regions of biomass burning seems to be limited by heterogeneous losses that accompany convective processes that ventilate the surface layer and the lower boundary layer (Thornton et al., 1996, TRACE-P data archive).

However, it should be noted that reduced sulfur species are also produced by industry. For example, DMS is used in petroleum refining and in petrochemical production processes to control the formation of coke and carbon monoxide. In addition, it is used to control dusting in steel mills. It is also used in a range of organic syntheses. It also has a use as a food flavoring component. It can also be oxidized by natural or artificial means to dimethyl sulfoxide (DMSO), which has several important solvent properties.

AX2.6.3 Field Studies Evaluating Emissions Inventories

Comparisons of emissions model predictions with observations have been performed in a number of environments. A number of studies of ratios of concentrations of CO to NO_x and NMOC to NO_x during the early 1990s in tunnels and ambient air (summarized in Air Quality Criteria for Carbon Monoxide (U.S. Environmental Protection Agency, 2000)) indicated that emissions of CO and NMOC were systematically underestimated in emissions inventories. However, the results of more recent studies have been mixed in this regard, with many studies showing agreement to within ±50% (U.S. Environmental Protection Agency, 2000). Improvements in many areas have resulted from the process of emissions model development, evaluation, and further refinement. It should be remembered that the conclusions from these reconciliation studies depend on the assumption that NO_x emissions are predicted correctly by emissions factor models. Roadside remote sensing data indicate that over 50% of NMHC and CO emissions are produced by less than about 10% of the vehicles (Stedman et al., 1991). These "super-emitters" are typically poorly maintained vehicles. Vehicles of any age engaged in off-cycle operations (e.g., rapid accelerations) emit much more than if operated in normal driving modes. Bishop and Stedman (1996) found that the most important variables governing CO emissions are fleet age and owner maintenance.

Emissions inventories for North America can be evaluated by comparison to measured long-term trends and or ratios of pollutants in ambient air. A decadal field study of ambient CO at a rural site in the Eastern U.S. (Hallock-Waters et al., 1999) indicates a downward trend

consistent with the downward trend in estimated emissions over the period 1988 to 1999 (U.S. Environmental Protection Agency, 1997), even when a global downward trend is accounted for. Measurements at two urban areas in the United States confirmed the decrease in CO emissions (Parrish et al., 2002). That study also indicated that the ratio of CO to NO_x emissions decreased by almost a factor of three over 12 years (such a downward trend was noted in AQCD 96). Emissions estimates (U.S. Environmental Protection Agency, 1997) indicate a much smaller decrease in this ratio, suggesting that NO_x emissions from mobile sources may be underestimated and/or increasing. Parrish et al. (2002) conclude that O_3 photochemistry in U.S. urban areas may have become more NO_x-limited over the past decade.

Pokharel et al. (2002) employed remotely sensed emissions from on-road vehicles and fuel use data to estimate emissions in Denver. Their calculations indicate a continual decrease in CO, HC, and NO emissions from mobile sources over the 6-year study period. Inventories based on the ambient data were 30 to 70% lower for CO, 40% higher for HC, and 40 to 80% lower for NO than those predicted by the MOBILE6 model.

Stehr et al. (2000) reported simultaneous measurements of CO, SO_2, and NO_y at an East Coast site. By taking advantage of the nature of mobile sources (they emit NO_x and CO but little SO_2) and power plants (they emit NO_x and SO_2 but little CO), the authors evaluated emissions estimates for the eastern United States. Results indicated that coal combustion contributes 25 to 35% of the total NO_x emissions in rough agreement with emissions inventories (U.S. Environmental Protection Agency, 1997).

Parrish et al. (1998) and Parrish and Fehsenfeld (2000) proposed methods to derive emission rates by examining measured ambient ratios among individual VOC, NO_x and NO_y. There is typically a strong correlation among measured values for these species because emission sources are geographically collocated, even when individual sources are different. Correlations can be used to derive emissions ratios between species, including adjustments for the impact of photochemical aging. Investigations of this type include correlations between CO and NO_y (e.g., Parrish et al., 1991), between individual VOC species and NO_y (Goldan et al., 1995, 1997, 2000) and between various individual VOC (Goldan et al., 1995, 1997; McKeen and Liu, 1993; McKeen et al., 1996). Buhr et al. (1992) derived emission estimates from principal component analysis (PCA) and other statistical methods. Many of these studies are summarized in Trainer et al. (2000), Parrish et al. (1998), and Parrish and Fehsenfeld (2000). Goldstein and Schade

(2000) also used species correlations to identify the relative impacts of anthropogenic and biogenic emissions. Chang et al. (1996, 1997) and Mendoza-Dominguez and Russell (2000, 2001) used the more quantative technique of inverse modeling to derive emission rates, in conjunction with results from chemistry-transport models.

AX2.7 METHODS USED TO CALCULATE CONCENTRATIONS OF NITROGEN OXIDES AND THEIR CHEMICAL INTERACTIONS IN THE ATMOSPHERE

Atmospheric chemistry and transport models are the major tools used to calculate the relations among O_3, other oxidants, and their precursors, the transport and transformation of air toxics, the production of secondary organic aerosol, the evolution of the particle size distribution, and the production and deposition of pollutants affecting ecosystems. Chemical transport models are driven by emissions inventories for primary species such as the precursors for O_3 and PM and by meterological fields produced by other numerical models. Emissions of precursor compounds can be divided into anthropogenic and natural source categories. Natural sources can be further divided into biotic (vegetation, microbes, animals) and abiotic (biomass burning, lightning) categories. However, the distinction between natural sources and anthropogenic sources is often difficult to make as human activities affect directly, or indirectly, emissions from what would have been considered natural sources during the preindustrial era. Emissions from plants and animals used in agriculture have been referred to as anthropogenic or natural in different applications. Wildfire emissions may be considered to be natural, except that forest management practices may have led to the buildup of fuels on the forest floor, thereby altering the frequency and severity of forest fires. Needed meteorological quantities such as winds and temperatures are taken from operational analyses, reanalyses, or circulation models. In most cases, these are off-line analyses, i.e., they are not modified by radiatively active species such as O_3 and particles generated by the model.

A brief overview of atmospheric chemistry-transport models is given in Section AX2.7.1. A discussion of emissions inventories of precursors used by these models is given in Section AX2.7.2. Uncertainties in emissions estimates have also been discussed in Air Quality Criteria for Particulate Matter (U.S. Environmental Protection Agency, 2004). Chemistry-transport

model evaluation and an evaluation of the reliability of emissions inventories are presented in Section AX2.7.4.

AX2.7.1 Chemistry-Transport Models

Atmospheric CTMs have been developed for application over a wide range of spatial scales ranging from neighborhood to global. Regional scale CTMs are used: 1) to obtain better understanding of the processes controlling the formation, transport, and destruction of gas-and particle-phase criteria and hazardous air pollutants; 2) to understand the relations between O_3 concentrations and concentrations of its precursors such as NO_x and VOCs, the factors leading to acid deposition, and hence to possible damage to ecosystems; and 3) to understand relations among the concentration patterns of various pollutants that may exert adverse health effects. Chemistry Transport Models are also used for determining control strategies for O_3 precursors. However, this application has met with varying degrees of success because of the highly nonlinear relations between O_3 and emissions of its precursors, and uncertainties in emissions, parameterizations of transport, and chemical production and loss terms. Uncertainties in meteorological variables and emissions can be large enough to lead to significant errors in developing control strategies (e.g., Russell and Dennis, 2000; Sillman et al., 1995).

Global scale CTMs are used to address issues associated with climate change, stratospheric ozone depletion, and to provide boundary conditions for regional scale models. CTMs include mathematical (and often simplified) descriptions of atmospheric transport, the transfer of solar radiation through the atmosphere, chemical reactions, and removal to the surface by turbulent motions and precipitation for pollutants emitted into the model domain. Their upper boundaries extend anywhere from the top of the mixing layer to the mesopause (about 80 km in height), to obtain more realistic boundary conditions for problems involving stratospheric dynamics. There is a trade-off between the size of the modeling domain and the grid resolution used in the CTM that is imposed by computational resources.

There are two major formulations of CTMs in current use. In the first approach, grid-based, or Eulerian, air quality models, the region to be modeled (the modeling domain) is subdivided into a three-dimensional array of grid cells. Spatial derivatives in the species continuity equations are cast in finite-difference there are also some finite-element models, but not many applications form over this grid, and a system of equations for the concentrations of all

the chemical species in the model are solved numerically at each grid point. Time dependent continuity (mass conservation) equations are solved for each species including terms for transport, chemical production and destruction, and emissions and deposition (if relevant), in each cell. Chemical processes are simulated with ordinary differential equations, and transport processes are simulated with partial differential equations. Because of a number of factors such as the different time scales inherent in different processes, the coupled, nonlinear nature of the chemical process terms, and computer storage limitations, all of the terms in the equations are not solved simultaneously in three dimensions. Instead, operator splitting, in which terms in the continuity equation involving individual processes are solved sequentially, is used. In the second CTM formulation, trajectory or Lagrangian models, a large number of hypothetical air parcels are specified as following wind trajectories. In these models, the original system of partial differential equations is transformed into a system of ordinary differential equations.

A less common approach is to use a hybrid Lagrangian/Eulerian model, in which certain aspects of atmospheric chemistry and transport are treated with a Lagrangian approach and others are treaded in a**n** Eulerian manner (e.g., Stein et al., 2000). Each approach has **its** their advantages and disadvantages. The Eulerian approach is more general in that it includes processes that mix air parcels and allows integrations to be carried out for long periods during which individual air parcels lose their identity. There are, however, techniques for including the effects of mixing in Lagrangian models such as FLEXPART (e.g., Zanis et al., 2003), ATTILA (Reithmeier and Sausen, 2002), and CLaMS (McKenna et al., 2002).

Regional Scale Chemistry Transport Models

Major modeling efforts within the U.S. Environmental Protection Agency center on the Community Multiscale Air Quality modeling system (CMAQ, Byun and Ching, 1999; Byun and Schere, 2006). A number of other modeling platforms using Lagrangian and Eulerian frameworks have been reviewed in the 96 AQCD for O_3 (U.S. EPA, 1997), and in Russell and Dennis (2000). The capabilities of a number of CTMs designed to study local- and regional-scale air pollution problems are summarized by Russell and Dennis (2000). Evaluations of the performance of CMAQ are given in Arnold et al. (2003), Eder and Y (2005), Appel et al. (2005), and Fuentes and Raftery (2005). The domain of CMAQ can extend from several hundred km to the hemispherical scale. In addition, both of these classes of models allow the resolution of the calculations over specified areas to vary. CMAQ is most often driven by the MM5 mesoscale

meteorological model (Seaman, 2000), though it may be driven by other meteorological models (e.g., RAMS). Simulations of O_3 episodes over regional domains have been performed with a horizontal resolution as low as 1 km, and smaller calculations over limited domains have been accomplished at even finer scales. However, simulations at such high resolutions require better parameterizations of meteorological processes such as boundary layer fluxes, deep convection and clouds (Seaman, 2000), and finer-scale emissions. Finer spatial resolution is necessary to resolve features such as urban heat island circulations; sea, bay, and land breezes; mountain and valley breezes, and the nocturnal low-level jet.

The most common approach to setting up the horizontal domain is to nest a finer grid within a larger domain of coarser resolution. However, there are other strategies such as the stretched grid (e.g., Fox-Rabinovitz et al., 2002) and the adaptive grid. In a stretched grid, the grid's resolution continuously varies throughout the domain, thereby eliminating any potential problems with the sudden change from one resolution to another at the boundary. Caution should be exercised in using such a formulation, because certain parameterizations that are valid on a relatively coarse grid scale (such as convection) may not be valid on finer scales. Adaptive grids are not fixed at the start of the simulation, but instead adapt to the needs of the simulation as it evolves (e.g., Hansen et al., 1994). They have the advantage that they can resolve processes at relevant spatial scales. However, they can be very slow if the situation to be modeled is complex. Additionally, if adaptive grids are used for separate meteorological, emissions, and photochemical models, there is no reason a priori why the resolution of each grid should match, and the gains realized from increased resolution in one model will be wasted in the transition to another model. The use of finer horizontal resolution in CTMs will necessitate finer-scale inventories of land use and better knowledge of the exact paths of roads, locations of factories, and, in general, better methods for locating sources and estimating their emissions.

The vertical resolution of these CTMs is variable, and usually configured to have higher resolution near the surface and decreasing aloft. Because the height of the boundary layer is of critical importance in simulations of air quality, improved resolution of the boundary layer height would likely improve air quality simulations. Additionally, current CTMs do not adequately resolve fine scale features such as the nocturnal low-level jet in part because little is known about the nighttime boundary layer.

CTMs require time-dependent, three-dimensional wind fields for the period of simulation. The winds may be either generated by a model using initial fields alone or with four-dimensional data assimilation to improve the model's performance, fields (i.e., model equations can be updated periodically or "nudged", to bring results into agreement with observations. Modeling efforts typically focus on simulations of several days' duration, the typical time scale for individual O_3 episodes, but there have been several attempts at modeling longer periods. For example, Kasibhatla and Chameides (2000) simulated a four-month period from May to September of 1995 using MAQSIP. The current trend in modeling applications is towards annual simulations. This trend is driven in part by the need to better understand observations of periods of high wintertime PM (e.g., Blanchard et al., 2002) and the need to simulate O_3 episodes occurring outside of summer.

Chemical kinetics mechanisms (a set of chemical reactions) representing the important reactions occurring in the atmosphere are used in CTMs to estimate the rates of chemical formation and destruction of each pollutant simulated as a function of time. Unfortunately, chemical mechanisms that explicitly treat the reactions of each individual reactive species are too computationally demanding to be incorporated into CTMs. For example, a master chemical mechanism includes approximately 10,500 reactions involving 3603 chemical species (Derwent et al., 2001). Instead, "lumped" mechanisms, that group compounds of similar chemistry together, are used. The chemical mechanisms used in existing photochemical O_3 models contain significant uncertainties that may limit the accuracy of their predictions; the accuracy of each of these mechanisms is also limited by missing chemistry. Because of different approaches to the lumping of organic compounds into surrogate groups, chemical mechanisms can produce somewhat different results under similar conditions. The CB-IV chemical mechanism (Gery et al., 1989), the RADM II mechanism (Stockwell et al., 1990), the SAPRC (e.g., Wang et al., 2000a,b; Carter, 1990) and the RACM mechanisms can be used in CMAQ. Jimenez et al. (2003) provide brief descriptions of the features of the main mechanisms in use and they compared concentrations of several key species predicted by seven chemical mechanisms in a box model simulation over 24 h. The average deviation from the average of all mechanism predictions for O_3 and NO over the daylight period was less than 20%, and was 10% for NO_2 for all mechanisms. However, much larger deviations were found for HNO_3, PAN, HO_2, H_2O_2, C_2H_4, and C_5H_8 (isoprene). An analysis for OH radicals was not presented. The large deviations

shown for most species imply differences between the calculated lifetimes of atmospheric species and the assignment of model simulations to either NO_x-limited or radical quantity limited regimes between mechanisms. Gross and Stockwell (2003) found small differences between mechanisms for clean conditions, with differences becoming more significant for polluted conditions, especially for NO_2 and organic peroxy radicals. They caution modelers to consider carefully the mechanisms they are using. Faraji et al. (2006) found differences of 40% in peak 1h O_3 in the Houston-Galveston-Brazoria area between simulations using SAPRAC and CB4. They attributed differences in predicted O_3 concentrations to differences in the mechanisms of oxidation of aromatic hydrocarbons.

CMAQ and other CTMs (e.g., PM-CAMx) incorporate processes and interactions of aerosol-phase chemistry (Mebust et al., 2003). There have also been several attempts to study the feedbacks of chemistry on atmospheric dynamics using meteorological models, like MM5 (e.g., Grell et al., 2000; Liu et al., 2001a; Lu et al., 1997; Park et al., 2001). This coupling is necessary to simulate accurately feedbacks such as may be caused by the heavy aerosol loading found in forest fire plumes (Lu et al., 1997; Park et al., 2001), or in heavily polluted areas. Photolysis rates in CMAQ can now be calculated interactively with model produced O_3, NO_2, and aerosol fields (Binkowski et al., 2007).

Spatial and temporal characterizations of anthropogenic and biogenic precursor emissions must be specified as inputs to a CTM. Emissions inventories have been compiled on grids of varying resolution for many hydrocarbons, aldehydes, ketones, CO, NH_3, and NO_x. Emissions inventories for many species require the application of some algorithm for calculating the dependence of emissions on physical variables such as temperature and to convert the inventories into formatted emission files required by a CTM. For example, preprocessing of emissions data for CMAQ is done by the SMOKE (Spare-Matrix Operator Kernel Emissions) system. For many species, information concerning the temporal variability of emissions is lacking, so long-term (e.g., annual or O_3-season) averages are used in short-term, episodic simulations. Annual emissions estimates are often modified by the emissions model to produce emissions more characteristic of the time of day and season. Significant errors in emissions can occur if an inappropriate time dependence or a default profile is used. Additional complexity arises in model calculations because different chemical mechanisms are based on different species, and inventories constructed for use with another mechanism must be adjusted to reflect

these differences. This problem also complicates comparisons of the outputs of these models because one chemical mechanism may produce some species not present in another mechanism yet neither may agree with the measurements.

In addition to wet deposition, dry deposition (the removal of chemical species from the atmosphere by interaction with ground-level surfaces) is an important removal process for pollutants on both urban and regional scales and must be included in CTMs. The general approach used in most models is the resistance in series method, in which where dry deposition is parameterized with a V_d, which is represented as $v_d = (r_a + r_b + r_c)^{-1}$ where r_a, r_b, and r_c represent the resistance due to atmospheric turbulence, transport in the fluid sublayer very near the elements of surface such as leaves or soil, and the resistance to uptake of the surface itself. This approach works for a range of substances, although it is inappropriate for species with substantial emissions from the surface or for species whose deposition to the surface depends on its concentration at the surface itself. The approach is also modified somewhat for aerosols: the terms r_b and r_c are replaced with a surface V_d to account for gravitational settling. In their review, Wesley and Hicks (2000) point out several shortcomings of current knowledge of dry deposition. Among those shortcomings are difficulties in representing dry deposition over varying terrain where horizontal advection plays a significant role in determining the magnitude of r_a and difficulties in adequately determining a V_d for extremely stable conditions such as those occurring at night (e.g., Mahrt, 1998). Under the best of conditions, when a model is exercised over a relatively small area where dry deposition measurements have been made, models still commonly show uncertainties at least as large as ±30% (e.g., Massman et al., 1994; Brook et al., 1996; Padro, 1996). Wesely and Hicks (2000) state that an important result of these comparisons is that the current level of sophistication of most dry deposition models is relatively low, and that deposition estimates therefore must rely heavily on empirical data. Still larger uncertainties exist when the surface features in the built environment are not well known or when the surface comprises a patchwork of different surface types, as is common in the eastern United States.

The initial conditions, i.e., the concentration fields of all species computed by a model, and the boundary conditions, i.e., the concentrations of species along the horizontal and upper boundaries of the model domain throughout the simulation must be specified at the beginning of the simulation. It would be best to specify initial and boundary conditions according to observations. However, data for vertical profiles of most species of interest are sparse. The

results of model simulations over larger, preferably global, domains can also be used. As may be expected, the influence of boundary conditions depends on the lifetime of the species under consideration and the time scales for transport from the boundaries to the interior of the model domain (Liu et al., 2001b).

Each of the model components described above has an associated uncertainty, and the relative importance of these uncertainties varies with the modeling application. The largest errors in photochemical modeling are still thought to arise from the meteorological and emissions inputs to the model (Russell and Dennis, 2000). Within the model itself, horizontal advection algorithms are still thought to be significant source of uncertainty (e.g., Chock and Winkler, 1994), though more recently, those errors are thought to have been reduced (e.g., Odman et al., 1996). There are also indications that problems with mass conservation continue to be present in photochemical and meteorological models (e.g., Odman and Russell, 1999); these can result in significant simulation errors. The effects of errors in initial conditions can be minimized by including several days "spin-up" time in a simulation to allow the model to be driven by emitted species before the simulation of the period of interest begins.

While the effects of poorly specified boundary conditions propagate through the model's domain, the effects of these errors remain undetermined. Because many meteorological processes occur on spatial scales which are smaller than the model grid spacing (either horizontally or vertically) and thus are not calculated explicitly, parameterizations of these processes must be used and these introduce additional uncertainty.

Uncertainty also arises in modeling the chemistry of O_3 formation because it is highly nonlinear with respect to NO_x concentrations. Thus, the volume of the grid cell into which emissions are injected is important because the nature of O_3 chemistry (i.e., O_3 production or titration) depends in a complicated way on the concentrations of the precursors and the OH radical as noted earlier. The use of ever-finer grid spacing allows regions of O_3 titration to be more clearly separated from regions of O_3 production. The use of grid spacing fine enough to resolve the chemistry in individual power-plant plumes is too demanding of computer resources for this to be attempted in most simulations. Instead, parameterizations of the effects of sub-grid-scale processes such as these must be developed; otherwise serious errors can result if emissions are allowed to mix through an excessively large grid volume before the chemistry step in a model calculation is performed. In light of the significant differences between atmospheric

chemistry taking place inside and outside of a power plant plume (e.g., Ryerson et al., 1998 and Sillman, 2000), inclusion of a separate, meteorological module for treating large, tight plumes is necessary. Because the photochemistry of O_3 and many other atmospheric species is nonlinear, emissions correctly modeled in a tight plume may be incorrectly modeled in a more dilute plume. Fortunately, it appears that the chemical mechanism used to follow a plume's development need not be as detailed as that used to simulate the rest of the domain, as the inorganic reactions are the most important in the plume see (e.g., Kumar and Russell, 1996). The need to include explicitly plume-in-grid chemistry only down to the level of the smallest grid disappears if one uses the adaptive grid approach mentioned previously, though such grids are more computationally intensive. The differences in simulations are significant because they can lead to significant differences in the calculated sensitivity of O_3 to its precursors (e.g., Sillman et al., 1995).

Because the chemical production and loss terms in the continuity equations for individual species are coupled, the chemical calculations must be performed iteratively until calculated concentrations converge to within some preset criterion. The number of iterations and the convergence criteria chosen also can introduce error.

Global Scale CTMs

The importance of global transport of O_3 and O_3 precursors and their contribution to regional O_3 levels in the United States is slowly becoming apparent. There are presently on the order of 20 three-dimensional global models that have been developed by various groups to address problems in tropospheric chemistry. These models resolve synoptic meteorology, O_3-NO_x-CO-hydrocarbon photochemistry, have parameterizations for wet and dry deposition, and parameterize sub-grid scale vertical mixing processes such as convection. Global models have proven useful for testing and advancing scientific understanding beyond what is possible with observations alone. For example, they can calculate quantities of interest that cannot be measured directly, such as the export of pollution from one continent to the global atmosphere or the response of the atmosphere to future perturbations to anthropogenic emissions.

Global simulations are typically conducted at a horizontal resolution of about 200 km^2. Simulations of the effects of transport from long-range transport link multiple horizontal resolutions from the global to the local scale. Finer resolution will only improve scientific understanding to the extent that the governing processes are more accurately described at that

scale. Consequently, there is a critical need for observations at the appropriate scales to evaluate the scientific understanding represented by the models.

During the recent IPCC-AR4 tropospheric chemistry study coordinated by the European Union project Atmospheric Composition Change: the European Network of excellence (ACCENT), 26 atmospheric CTMs were used to estimate the impacts of three emissions scenarios on global atmospheric composition, climate, and air quality in 2030 (Dentener et al., 2006a). All models were required to use anthropogenic emissions developed at IIASA (Dentener et al., 2005) and GFED version 1 biomass burning emissions (van der Werf et al., 2003) as described in Stevenson et al. (2006). The base simulations from these models were evaluated against a suite of present-day observations. Most relevant to this assessment report are the evaluations with ozone and NO_2, and for nitrogen and sulfur deposition (Stevenson et al., 2006; van Noije et al., 2006; Dentener et al., 2006a), which are summarized briefly below.

An analysis of the standard deviation of zonal mean and tropospheric column O_3 reveals large inter-model variability in the tropopause region and throughout the polar troposphere, likely reflecting differences in model tropopause levels and the associated stratospheric injection of O_3 to the troposphere (Stevenson et al., 2006). Ozone distributions in the tropics also exhibit large standard deviations (~30%), particularly as compared to the mid-latitudes (~20%), indicating larger uncertainties in the processes that influence ozone in the tropics: deep tropical convection, lightning NO_x, isoprene emissions and chemistry, and biomass burning emissions (Stevenson et al., 2006).

Stevenson et al., (2006) found that the model ensemble mean (MEM) typically captures the observed seasonal cycles to within one standard deviation. The largest discrepancies between the MEM and observations include: (1) an underestimate of the amplitude of the seasonal cycle at 30°-90°N with a 10 ppbv overestimate of winter ozone, possibly due to the lack of a seasonal cycle in anthropogenic emissions or to shortcomings in the stratospheric influx of O_3, and (2) an overestimate of O_3 throughout the northern tropics. However, the MEM was found to capture the observed seasonal cycles in the Southern Hemisphere, suggesting that the models adequately represent biomass burning and natural emissions.

The mean present-day global ozone budget across the current generation of CTMs differs substantially from that reported in the IPCC TAR, with a 50% increase in the mean chemical production (to 5100 Tg O_3 yr^{-1}), a 30% increase in the chemical and deposition loss terms (to

4650 and 1000 Tg O_3 yr^{-1}, respectively) and a 30% decrease in the mean stratospheric input flux (to 550 Tg O_3 yr^{-1}) (Stevenson et al., 2006). The larger chemical terms as compared to the IPCC TAR are attributed mainly to higher NO_x (as well as an equatorward shift in distribution) and isoprene emissions, although more detailed NMHC schemes and/or improved representations of photolysis, convection, and stratospheric-tropospheric exchange may also contribute (Stevenson et al., 2006).

A subset of 17 of the 26 models used in the Stevenson et al. (2006) study was used to compare with three retrievals of NO_2 columns from the GOME instrument (van Noije et al., 2006) for the year 2000. The higher resolution models reproduce the observed patterns better, and the correlation among simulated and retrieved columns improved for all models when simulated values are smoothed to a $5° \times 5°$ grid, implying that the models do not accurately reproduce the small-scale features of NO_2 (van Noije et al., 2006). Van Noije et al. (2006) suggest that variability in simulated NO_2 columns may reflect a model differences in OH distributions and the resulting NO_x lifetimes, as well as differences in vertical mixing which strongly affect partitioning between NO and NO_2. Overall, the models tend to underestimate concentrations in the retrievals in industrial regions (including the eastern United States) and overestimate them in biomass burning regions (van Noije et al., 2006).

Over the eastern United States, and industrial regions more generally, the spread in absolute column abundances is generally larger among the retrievals than among the models, with the discrepancy among the retrievals particularly pronounced in winter (van Noije et al., 2006), suggesting that the models are biased low, or that the European retrievals may be biased high as the Dalhousie/SAO retrieval is closer to the model estimates. The lack of seasonal variability in fossil fuel combustion emissions may contribute to a wintertime model underestimate (van Noije et al., 2006) that is manifested most strongly over Asia. In biomass burning regions, the models generally reproduce the timing of the seasonal cycle of the retrievals, but tend to overestimate the seasonal cycle amplitude, partly due to lower values in the wet season, which may reflect an underestimate in wet season soil NO emissions (van Noije et al., 2006, Jaegle et al., 2004, 2005).

Deposition in Global CTMs

Both wet and dry deposition are highly parameterized in global CTMs. While all current models implement resistance schemes for dry deposition, the generated V_d generated from different models can vary highly across terrains (Stevenson et al., 2006). The accuracy of wet deposition in global CTMs is tied to spatial and temporal distribution of model precipitation and the treatment of chemical scavenging. Dentener et al. (2006b) compared wet deposition across 23 models with available measurements around the globe. Figures AX2-13 and AX2-14 below extract the results of a comparison of the 23-model mean versus observations from Dentener et al. (2006b) over the eastern United States for nitrate and sulfate deposition, respectively. The mean model results are strongly correlated with the observations ($r > 0.8$), and usually capture the magnitude of wet deposition to within a factor of 2 over the eastern United States (Dentener et al., 2006b). Dentener et al. (2006b) conclude that 60-70% of the participating models capture the measurements to within 50% in regions with quality controlled observations. This study then identified world regions receiving >1000 mg (N) m^{-2} yr^{-1} (the "critical load") and found that 20% of the natural vegetation (non-agricultural) in the United States is exposed to nitrogen deposition in excess of the critical load threshold (Dentener et al., 2006b).

Modeling the Effects of Convection

The effects of deep convection can be simulated using cloud-resolving models, or in regional or global models in which the convection is parameterized. The Goddard Cumulus Ensemble (GCE) model (Tao and Simpson, 1993) has been used by Pickering et al. (1991; 1992a,b; 1993; 1996), Scala et al. (1990) and Stenchikov et al. (1996) in the analysis of convective transport of trace gases. The cloud model is nonhydrostatic and contains a detailed representation of cloud microphysical processes. Two- and three-dimensional versions of the model have been applied in transport analyses. The initial conditions for the model are usually from a sounding of temperature, water vapor and winds representative of the region of storm development. Model-generated wind fields can be used to perform air parcel trajectory analyses and tracer advection calculations.

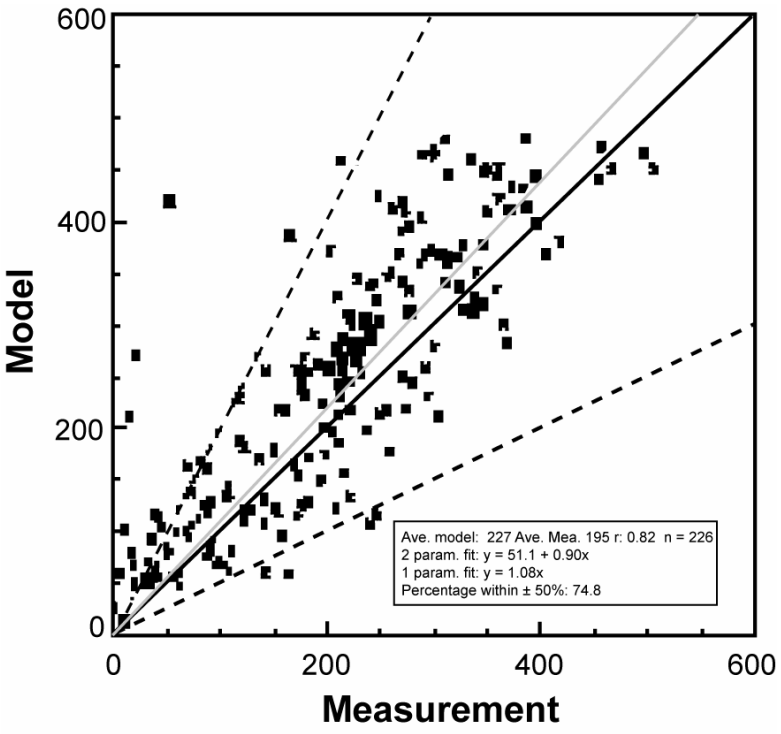


Figure AX2-13. Scatter plot of total nitrate (HNO_3 plus aerosol nitrate) wet deposition (mg(N)m $2yr^{-1}$) of the mean model versus measurements for the North American Deposition Program (NADP) network. Dashed lines indicate factor of 2. The gray line is the result of a linear regression fitting through 0.

Source: Dentener et al. (2006b).

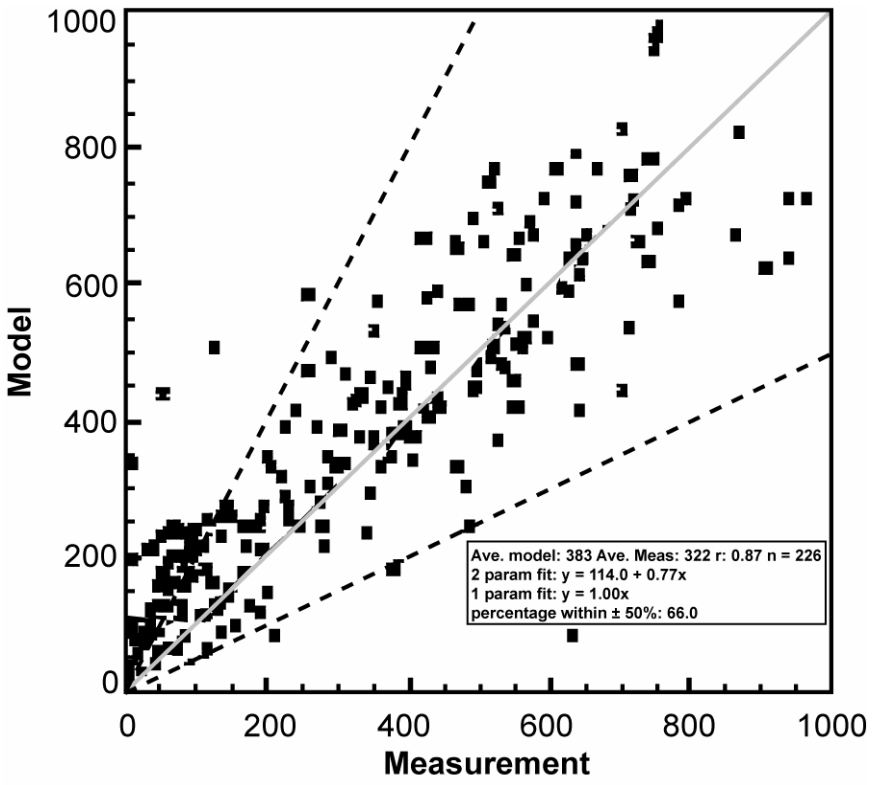


Figure AX2-14. Same as Figure AX2-13 but for sulfate wet deposition ($mg(S)m^{-2}yr^{-1}$).

Source: Dentener et al. (2006b).

Such methods were used by Pickering et al. (1992b) to examine transport of urban plumes by deep convection. Transport of an Oklahoma City plume by the 10-11 June 1985 PRE-STORM squall line was simulated with the 2-D GCE model. This major squall line passed over the Oklahoma City metropolitan area, as well as more rural areas to the north. Chemical observations ahead of the squall line were conducted by the PRE-STORM aircraft. In this event, forward trajectories from the boundary layer at the leading edge of the storm showed that almost 75% of the low-level inflow was transported to altitudes exceeding 8 km. Over 35% of the air parcels reached altitudes over 12 km. Tracer transport calculations were performed for CO, NO_x, O_3, and hydrocarbons. Rural boundary layer NO_x was only 0.9 ppbv, whereas the urban plume contained about 3 ppbv. In the rural case, mixing ratios of 0.6 ppbv were transported up to 11 km. Cleaner air descended at the rear of the storm lowering NO_x at the surface from 0.9 to

0.5 ppbv. In the urban plume, mixing ratios in the updraft core reached 1 ppbv between 14 and 15 km. At the surface, the main downdraft lowered the NO_x mixing ratios from 3 to 0.7 ppbv.

Regional chemical transport models have been used for applications such as simulations of photochemical O_3 production, acid deposition, and fine PM. Walcek et al. (1990) included a parameterization of cloud-scale aqueous chemistry, scavenging, and vertical mixing in the chemistry model of Chang et al. (1987). The vertical distribution of cloud microphysical properties and the amount of sub-cloud-layer air lifted to each cloud layer are determined using a simple entrainment hypothesis (Walcek and Taylor, 1986). Vertically integrated O_3 formation rates over the northeast U. S. were enhanced by ~50% when the in-cloud vertical motions were included in the model.

Wang et al. (1996) simulated the 10-11 June 1985 PRE-STORM squall line with the NCAR/Penn State Mesoscale Model (MM5; Grell et al., 1994; Dudhia, 1993). Convection was parameterized as a sub-grid-scale process in MM5 using the Kain Fritsch (1993) scheme. Mass fluxes and detrainment profiles from the convective parameterization were used along with the 3-D wind fields in CO tracer transport calculations for this convective event.

Convective transport in global chemistry and transport models is treated as a sub-grid-scale process that is parameterized typically using cloud mass flux information from a general circulation model or global data assimilation system. While GCMs can provide data only for a "typical" year, data assimilation systems can provide "real" day-by-day meteorological conditions, such that CTM output can be compared directly with observations of trace gases. The NASA Goddard Earth Observing System Data Assimilation System (GEOS-1 DAS and successor systems; Schubert et al., 1993; Bloom et al., 1996; Bloom et al., 2005) provides archived global data sets for the period 1980 to present, at $2^o \times 2.5^o$ or better resolution with 20 layers or more in the vertical. Deep convection is parameterized with the Relaxed Arakawa-Schubert scheme (Moorthi and Suarez, 1992) in GEOS-1 and GEOS-3 and with the Zhang and McFarlane (1995) scheme in GEOS-4. Pickering et al. (1995) showed that the cloud mass fluxes from GEOS-1 DAS are reasonable for the 10-11 June 1985 PRE-STORM squall line based on comparisons with the GCE model (cloud-resolving model) simulations of the same storm. In addition, the GEOS-1 DAS cloud mass fluxes compared favorably with the regional estimates of convective transport for the central U. S. presented by Thompson et al. (1994). However, Allen et al. (1997) have shown that the GEOS-1 DAS overestimates the amount and

frequency of convection in the tropics and underestimates the convective activity over midlatitude marine storm tracks.

Global models with parameterized convection and lightning have been run to examine the roles of these processes over North America. Lightning contributed 23% of upper tropospheric NO_y over the SONEX region according to the UMD-CTM modeling analysis of Allen et al. (2000). During the summer of 2004 the NASA Intercontinental Chemical Transport Experiment - North America (INTEX-NA) was conducted primarily over the eastern two-thirds of the United States, as a part of the International Consortium for Atmospheric Research on Transport and Transformation (ICARTT). Deep convection was prevalent over this region during the experimental period. Cooper et al. (2006) used a particle dispersion model simulation for NO_x to show that 69-84% of the upper tropospheric O_3 enhancement over the region in Summer 2004 was due to lightning NO_x. The remainder of the enhancement was due to convective transport of O_3 from the boundary layer or other sources of NO_x. Hudman et al. (2007) used a GEOS-Chem model simulation to show that lightning was the dominant source of upper tropospheric NO_x over this region during this period. Approximately 15% of North American boundary layer NO_x emissions were shown to have been vented to the free troposphere over this region based on both the observations and the model.

AX2.7.2 CTM Evaluation

The comparison of model predictions with ambient measurements represents a critical task for establishing the accuracy of photochemical models and evaluating their ability to serve as the basis for making effective control strategy decisions. The evaluation of a model's performance, or its adequacy to perform the tasks for which it was designed can only be conducted within the context of measurement errors and artifacts. Not only are there analytical problems, but there are also problems in assessing the representativeness of monitors at ground level for comparison with model values which represent typically an average over the volume of a grid box.

Evaluations of CMAQ are given in Arnold et al. (2003) and Fuentes and Raftery (2005). Discrepancies between model predictions and observations can be used to point out gaps in current understanding of atmospheric chemistry and to spur improvements in parameterizations of atmospheric chemical and physical processes. Model evaluation does not merely involve a

straightforward comparison between model predictions and the concentration field of the pollutant of interest. Such comparisons may not be meaningful because it is difficult to determine if agreement between model predictions and observations truly represents an accurate treatment of physical and chemical processes in the CTM or the effects of compensating errors in complex model routines. Ideally, each of the model components (emissions inventories, chemical mechanism, meteorological driver) should be evaluated individually. However, this is rarely done in practice.

Chemical transport models for O_3 formation at the urban/regional scale have traditionally been evaluated based on their ability to simulate correctly O_3. A series of performance statistics that measure the success of individual model simulations to represent the observed distribution of ambient O_3, as represented by a network of surface measurements at the urban scale were recommended by the U.S. Environmental Protection Agency (U.S. EPA, 1991; see also Russell and Dennis, 2000). These statistics consist of the following:

- Unpaired peak O_3 concentration within a metropolitan region (typically for a single day).
- Normalized bias equal to the summed difference between model and measured hourly concentrations divided by the sum of measured hourly concentrations.
- Normalized gross error, equal to the summed unsigned (absolute value) difference between model and measured hourly concentrations divided by the sum of measured hourly concentrations.

Unpaired peak prediction accuracy, A_u;

$$A_u = \frac{C_p(x,t)_{max} - C_o(x',t')_{max}}{C_o(x',t')_{max}} * 100\%, \qquad \text{(AX2-48)}$$

Normalized bias, D;

$$D = \frac{1}{N} \sum_{i=1}^{N} \frac{\{C_p(x_i,t) - C_o(x_i,t)\}}{C_o(x_i,t)}, t = 1,24. \qquad \text{(AX2-49)}$$

Gross error, E_d (for hourly observed values of O_3 >60 ppb)

$$E_d = \frac{1}{N}\sum_{i=1}^{N}\frac{\left| C_p(x_i,t) - C_o(x_i,t) \right|}{C_o(x_i,t)}, t = 1,24. \quad \text{(AX2-50)}$$

The following performance criteria for regulatory models were recommended in U.S. Environmental Protection Agency (1991): unpaired peak O_3 to within ±15% or ±20%; normalized bias within ± 5% to ± 15%; and normalized gross error less than 30% to 35%, but only when O_3 the concentration >60 ppb. This can lead to difficulties in evaluating model performance since nighttime and diurnal cycles are ignored. A major problem with this method of model evaluation is that it does not provide any information about the accuracy of O_3-precursor relations predicted by the model. The process of O_3 formation is sufficiently complex that models can predict O_3 correctly without necessarily representing the O_3 formation process properly. If the O_3 formation process is incorrect, then the modeled source-receptor relations will also be incorrect.

Studies by Sillman et al. (1995, 2003), Reynolds et al. (1996) and Pierce et al. (1998) have identified instances in which different model scenarios can be created with very different O_3-precursor sensitivity, but without significant differences in the predicted O_3 fields. Figures AX2-15a,b provides an example. Referring to the O_3-NO_x-VOC isopleth plot (Figure AX2-16), it can be seen that similar O_3 concentrations can be found for photochemical conditions that have very different sensitivity to NO_x and VOCs.

Global-scale CTMs have generally been evaluated by comparison with measurements for a wide array of species, rather than just for O_3 (e.g., Wang et al., 1998; Emmons et al., 2000; Bey et al., 2001; Hess, 2001; Fiore et al., 2002). These have included evaluation of major primary species (NO_x, CO, and selected VOCs) and an array of secondary species (HNO_3, PAN, H_2O_2) that are often formed concurrently with O_3. Models for urban and regional O_3 have also been evaluated against a broader ensemble of measurements in a few cases, often associated with measurement intensives (e.g., Jacobson et al., 1996; Lu et al., 1997; Sillman et al., 1998). The results of a comparison between observed and computed concentrations from Jacobson et al. (1996) for the Los Angeles Basin are shown in Figures AX2-17a,b.

The highest concentrations of primary species usually occur in close proximity to emission sources (typically in urban centers) and at times when dispersion rates are low. The diurnal cycle includes high concentrations at night, with maxima during the morning rush hour, and low concentrations during the afternoon (Figure AX2-17a). The afternoon minima are

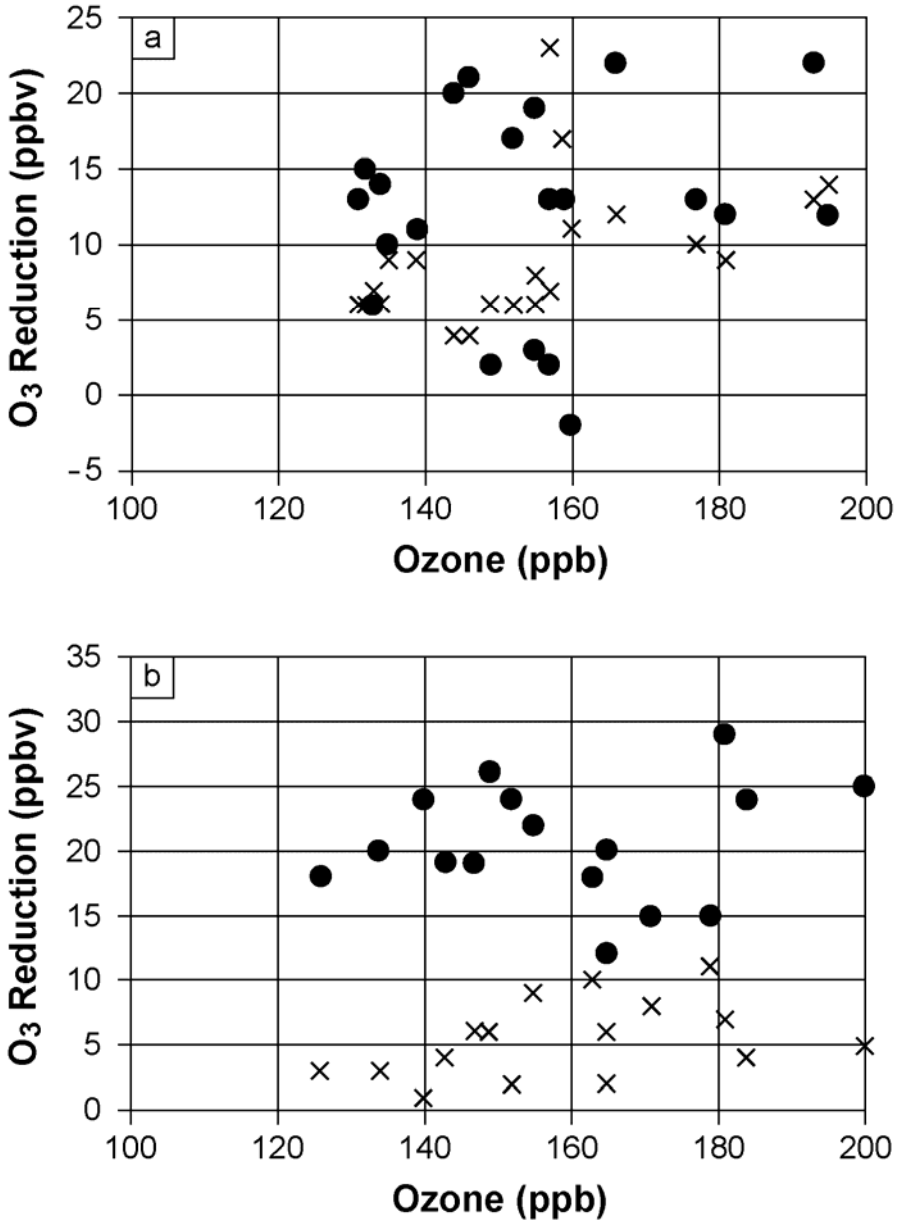


Figure AX2-15a,b. Impact of model uncertainty on control strategy predictions for O_3 for two days (August 10a and 11b, 1992) in Atlanta, GA. The figures show the predicted reduction in peak O_3 resulting from 35% reductions in anthropogenic VOC emissions (crosses) and from 35% reductions in NO_x (solid circles) in a series of model scenarios with varying base case emissions, wind fields, and mixed layer heights.

Source: Results are plotted from tabulated values published in Sillman et al. (1995, 1997).

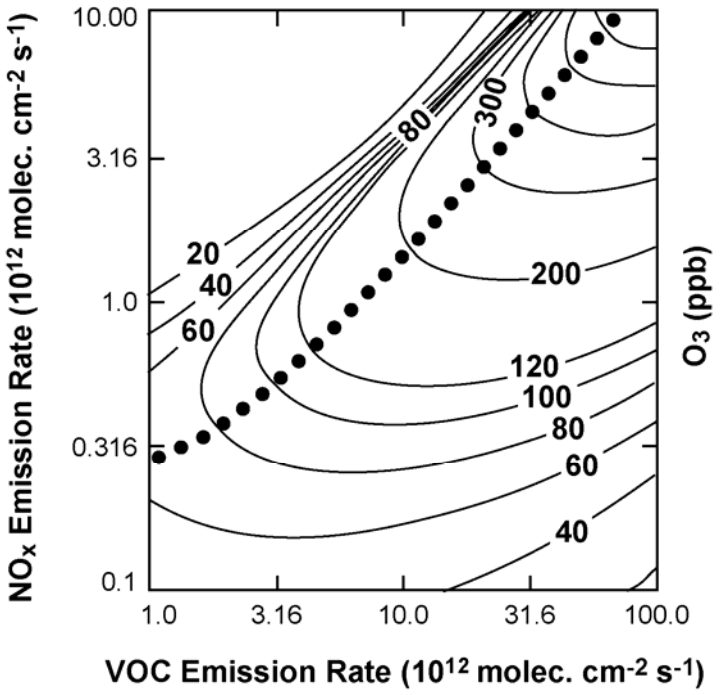


Figure AX2-16. **Ozone isopleths (ppb) as a function of the average emission rate for NO_x and VOC (10^{12} molec. cm^{-2} s^{-1}) in zero dimensional box model calculations. The isopleths (solid lines) represent conditions during the afternoon following 3-day calculations with a constant emission rate, at the hour corresponding to maximum O_3. The ridge line (shown by solid circles) lies in the transition from NO_x-saturated to NO_x-limited conditions.**

driven by the much greater rate of vertical mixing at that time. Primary species also show a seasonal maximum during winter, and are often high during fog episodes in winter when vertical mixing, is suppressed. By contrast, secondary species such as O_3 are typically highest during the afternoon (the time of greatest photochemical activity), on sunny days and during summer.

During these conditions, concentrations of primary species may be relatively low. Strong correlations between primary and secondary species are generally observed only in downwind rural areas where all anthropogenic species are simultaneously elevated. The difference in the diurnal cycles of primary species (CO, NO_x and ethane) and secondary species (O_3, PAN, and HCHO) is evident in Figure AX2-17b.

Models for urban and regional chemistry have been evaluated less extensively than global-scale models in part because the urban/regional context presents a number of difficult

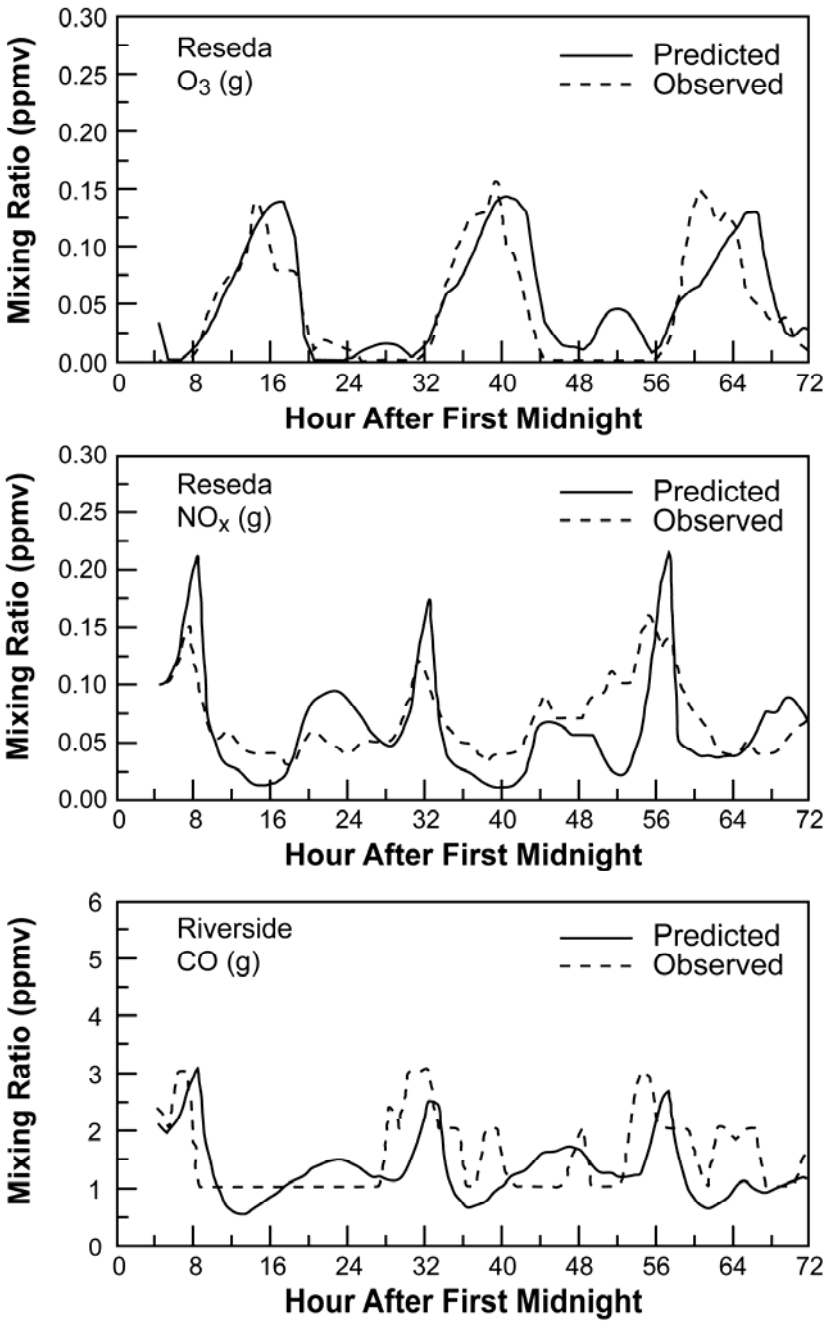


Figure AX2-17a. Time series for measured gas-phase species in comparison with results from a photochemical model. The dashed lines represent measurements, and solid lines represent model predictions (in parts per million, ppmv) for August 26–28, 1988 at sites in southern California. The horizontal axis represents hours past midnight, August 25. Results represent O_3 and NO_x at Reseda, and CO at Riverside.

Source: Jacobson et al. (1996).

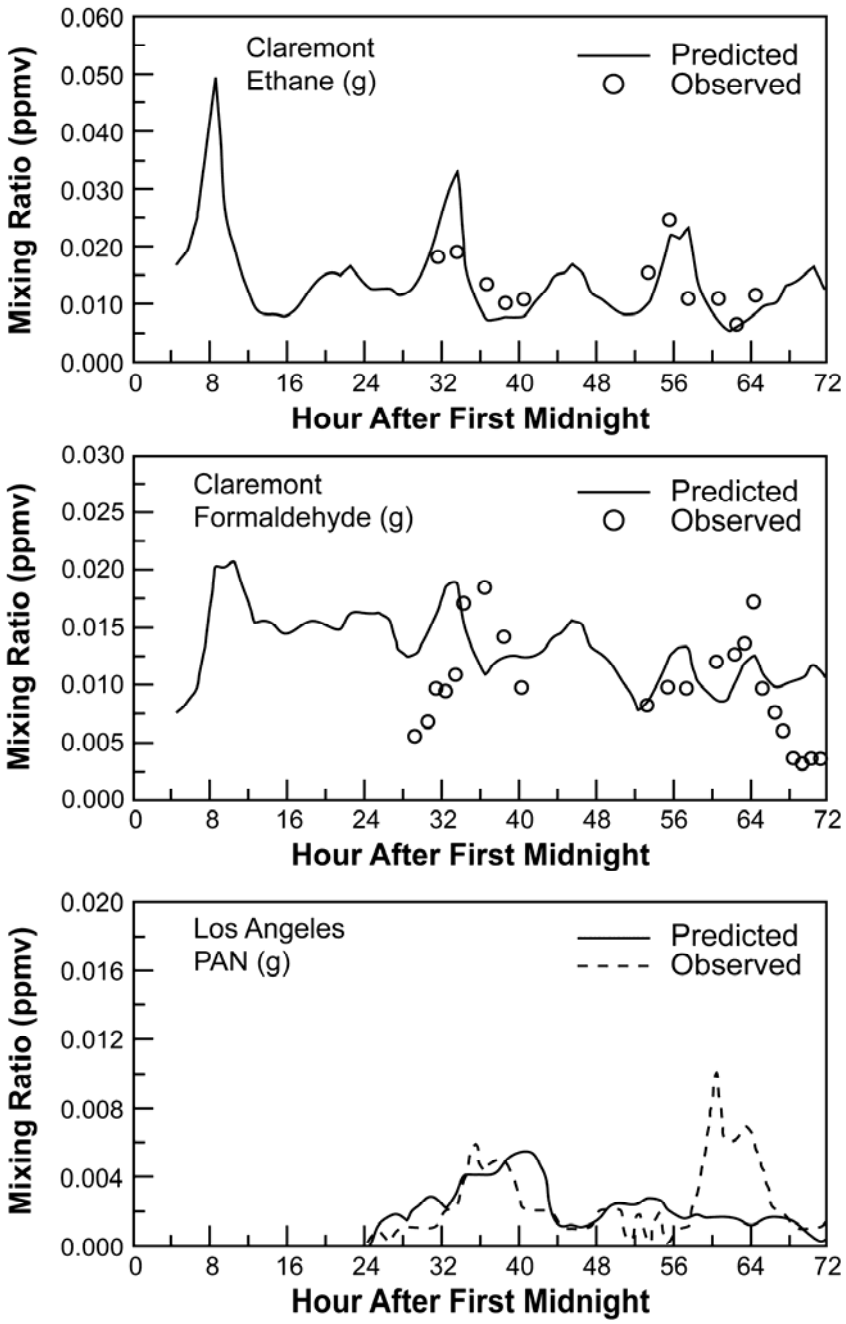


Figure AX2-17b. Time series for measured gas-phase species in comparison with results from a photochemical model. The circles represent measurements, and solid lines represent model predictions (in parts per million, ppmv) for August 26–28, 1988 at sites in southern California. The horizontal axis represents hours past midnight, August 25. Results represent ethane and formaldehyde at Claremont, and PAN at Los Angeles.

Source: Jacobson et al. (1996).

challenges. Global-scale models typically represent continental-scale events and can be evaluated effectively against a sparse network of measurements. By contrast, urban/regional models are critically dependent on the accuracy of local emission inventories and event-specific meteorology, and must be evaluated separately for each urban area that is represented.

The evaluation of urban/regional models is also limited by the availability of data. Measured NO_x and speciated VOC concentrations are widely available through the EPA PAMs network, but questions have been raised about the accuracy of those measurements and the data have not yet been analyzed thoroughly. Evaluation of urban/regional models versus measurements has generally relied on results from a limited number of field studies in the United States. Short-term, research-grade measurements for species relevant to O_3 formation, including VOCs, NO_x, PAN, HNO_3, and H_2O_2 are also available at selected rural and remote sites (e.g., Daum et al., 1990, 1996; Martin et al., 1997; Young et al., 1997; Thompson et al., 2000; Hoell et al., 1997, 1999; Fehsenfeld et al., 1996a; Emmons et al., 2000; Hess, 2001; Carroll et al., 2001). The equivalent measurements are available for some polluted rural sites in the eastern United States, but only at a few urban locations (Meagher et al., 1998; Hübler et al., 1998; Kleinman et al., 2000, 2001; Fast et al., 2002; new SCAQS-need reference). Extensive measurements have also been made in Vancouver (Steyn et al., 1997) and in several European cities (Staffelbach et al., 1997; Prévôt et al., 1997, Dommen et al., 1999; Geyer et al., 2001; Thielman et al., 2001; Martilli et al., 2002; Vautard et al., 2002).

The results of straightforward comparisons between observed and predicted concentrations of O_3 can be misleading because of compensating errors, although this possibility is diminished when a number of species are compared. Ideally, each of the main modules of a CTM system (for example, the meteorological model and the chemistry and radiative transfer routines) should be evaluated separately. However, this is rarely done in practice. To better indicate how well physical and chemical processes are being represented in the model, comparisons of relations between concentrations measured in the field and concentrations predicted by the model can be made. These comparisons could involve ratios and correlations between species. For example, correlation coefficients could be calculated between primary species as a means of evaluating the accuracy of emission inventories or between secondary species as a means of evaluating the treatment of photochemistry in the model. In addition, spatial relations involving individual species (correlations, gradients) can also be used as a means

of evaluating the accuracy of transport parameterizations. Sillman and He (2002) examined differences in correlation patterns between O_3 and NO_z in Los Angeles, CA, Nashville, TN, and various sites in the rural United States. Model calculations (Figure AX2-18) show differences in correlation patterns associated with differences in the sensitivity of O_3 to NO_x and VOCs. Primarily NO_x-sensitive (NO_x-limited) areas in models show a strong correlation between O_3 and NO_z with a relatively steep slope, while primarily VOC-sensitive (NO_x-saturated) areas in models show lower O_3 for a given NO_z and a lower O_3-NO_z slope. They found that differences found in measured data ensembles were matched by predictions from chemical transport models.

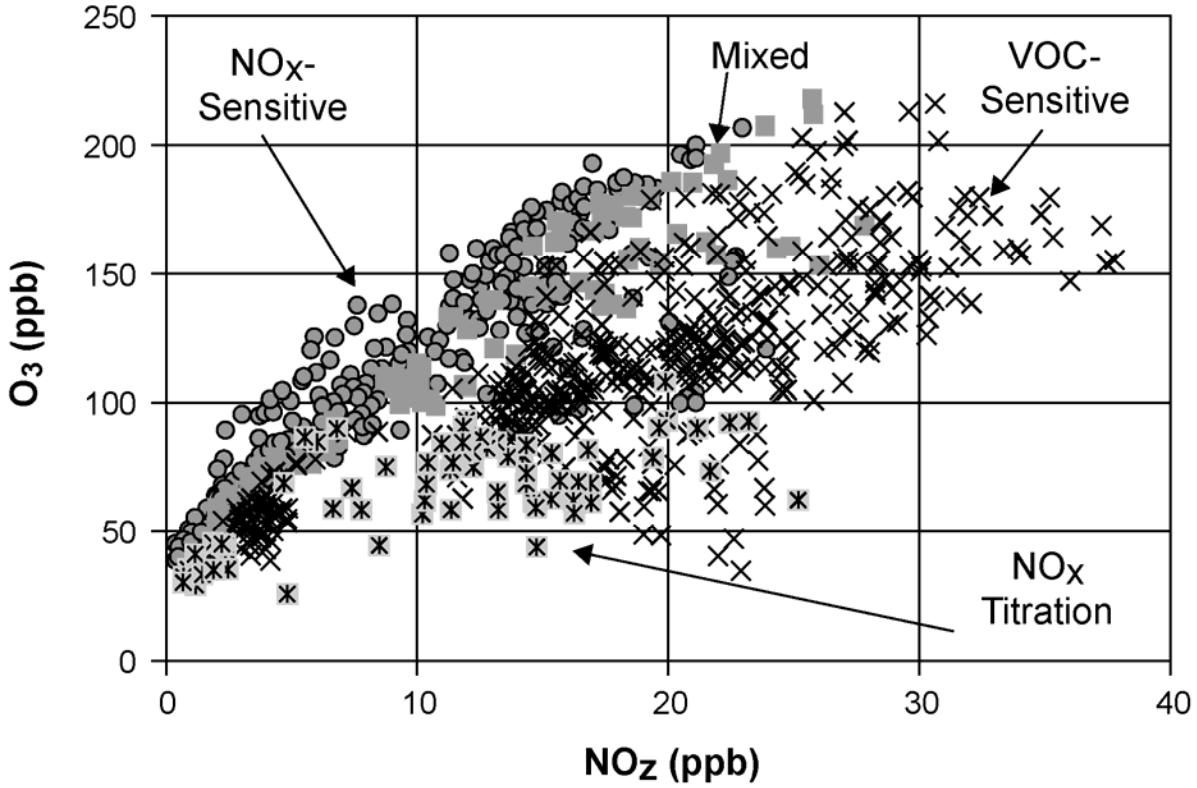


Figure AX2-18. Correlations for O_3 versus NO_z (NO_y–NO_x) in ppb from chemical transport models for the northeast corridor, Lake Michigan, Nashville, the San Joaquin Valley, and Los Angeles. Each location is classified as NO_x-limited or NO_x-sensitive (circles), NO_x-saturated or VOC-sensitive (crosses), mixed or with near-zero sensitivity (squares), and dominated by NO_x titration (asterisks) based on the model response to reduced NO_x and VOC.

Source: Sillman and He (2002).

Measurements in rural areas in the eastern U.S. show differences in the pattern of correlations for O_3 versus NO_z between summer and autumn (Jacob et al., 1995; Hirsch et al., 1996), corresponding to the transition from NO_x-limited to NO_x-saturated patterns, a feature which is also matched by CTMs.

The difference in correlations between secondary species in NO_x-limited to NO_x-saturated environments can also be used to evaluate the accuracy of model predictions in individual applications. Figures AX2-19a and AX2-19b show results for two different model scenarios for Atlanta. As shown in the figures, the first model scenario predicts an urban plume with high NO_y and O_3 formation apparently suppressed by high NO_y. Measurements show much lower NO_y in the Atlanta plume. This error was especially significant because the model locations sensitive to NO_x. The second model scenario (with primarily NO_x-sensitive conditions) shows much better agreement with measured values. Figure AX2-20a,b shows model-measurement comparisons for secondary species in Nashville, showing better agreement with measured with conditions. Greater confidence in the predictions made by CTMs will be gained by the application of techniques such as these on a more routine basis.

The ability of chemical mechanisms to calculate the concentrations of free radicals under atmospheric conditions was tested in the Berlin Ozone Experiment, BERLIOZ (Volz-Thomas et al., 2003) during July and early August at a site located about 50 km NW of Berlin. (This location was chosen because O_3 episodes in central Europe are often associated with SE winds.)

Concentrations of major compounds such as O_3, hydrocarbons, etc., were fixed at observed values. In this regard, the protocol used in this evaluation is an example of an observationally high NO_y were not sensitive to NO_x, while locations with lower NO_y were primarily based method. Figure AX2-21 compares the concentrations of RO_2, HO_2, and OH radicals predicted by RACM and MCM with observations made by the laser-induced fluorescence (LIF) technique and by matrix isolation ESR spectroscopy (MIESR). Also shown are the production rates of O_3 calculated using radical concentrations predicted by the mechanisms and those obtained by measurements, and measurements of NO_x concentrations. As can be seen, there is good agreement between measurements of RO_2, HO_2, OH, radicals with values predicted by both mechanisms at high concentrations of NO_x (>10 ppb). However, at lower NO_x concentrations, both mechanisms substantially overestimate OH concentrations and

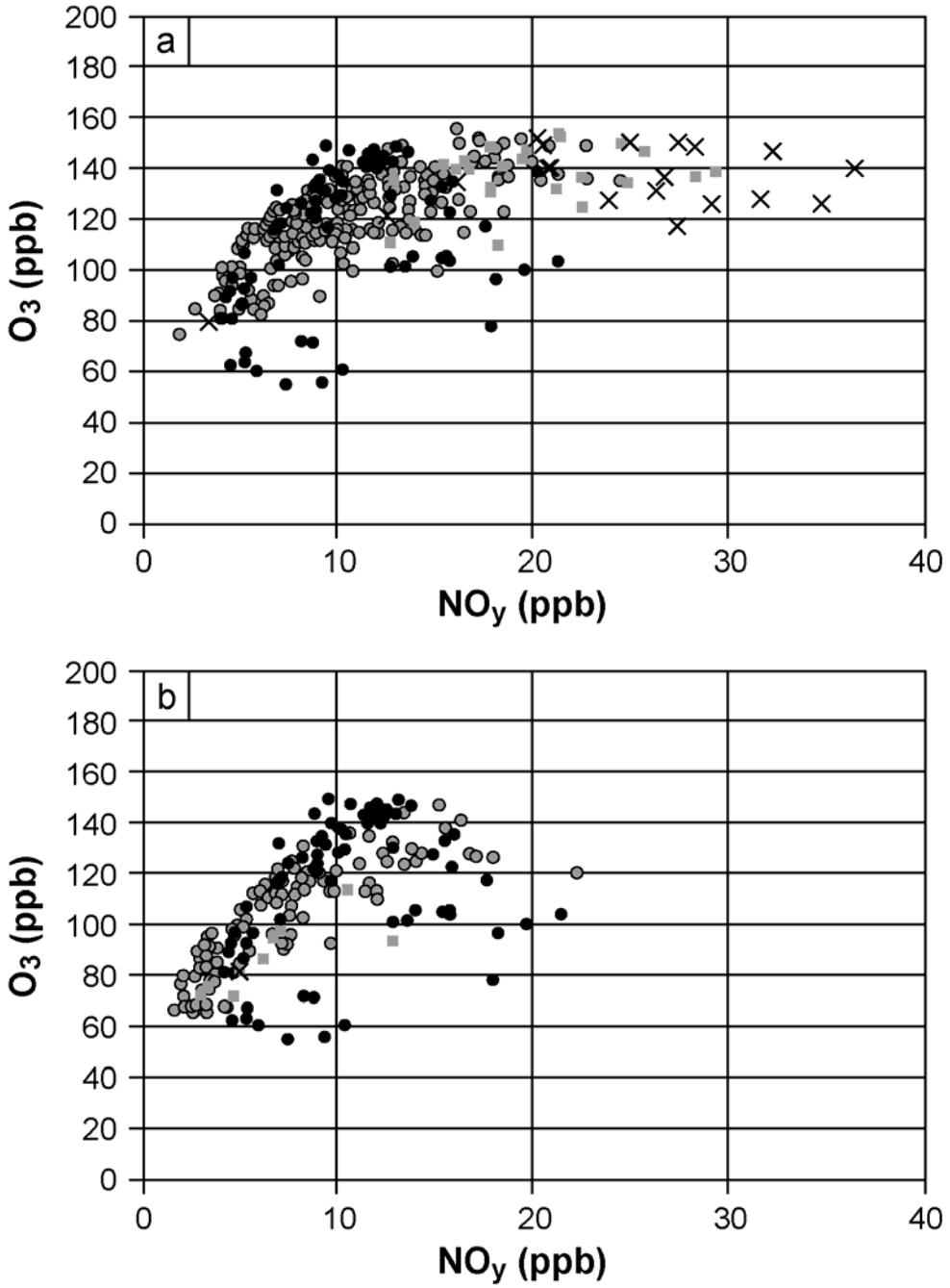


Figure AX2-19a,b. Evaluation of model versus measured O_3 versus NO_y for two model scenarios for Atlanta. The model values are classified as NO_x- limited (circles), NO_x-saturated (crosses), or mixed or with low sensitivity to NO_x (squares). Diamonds represent aircraft measurements.

Source: Sillman et al. (1997).

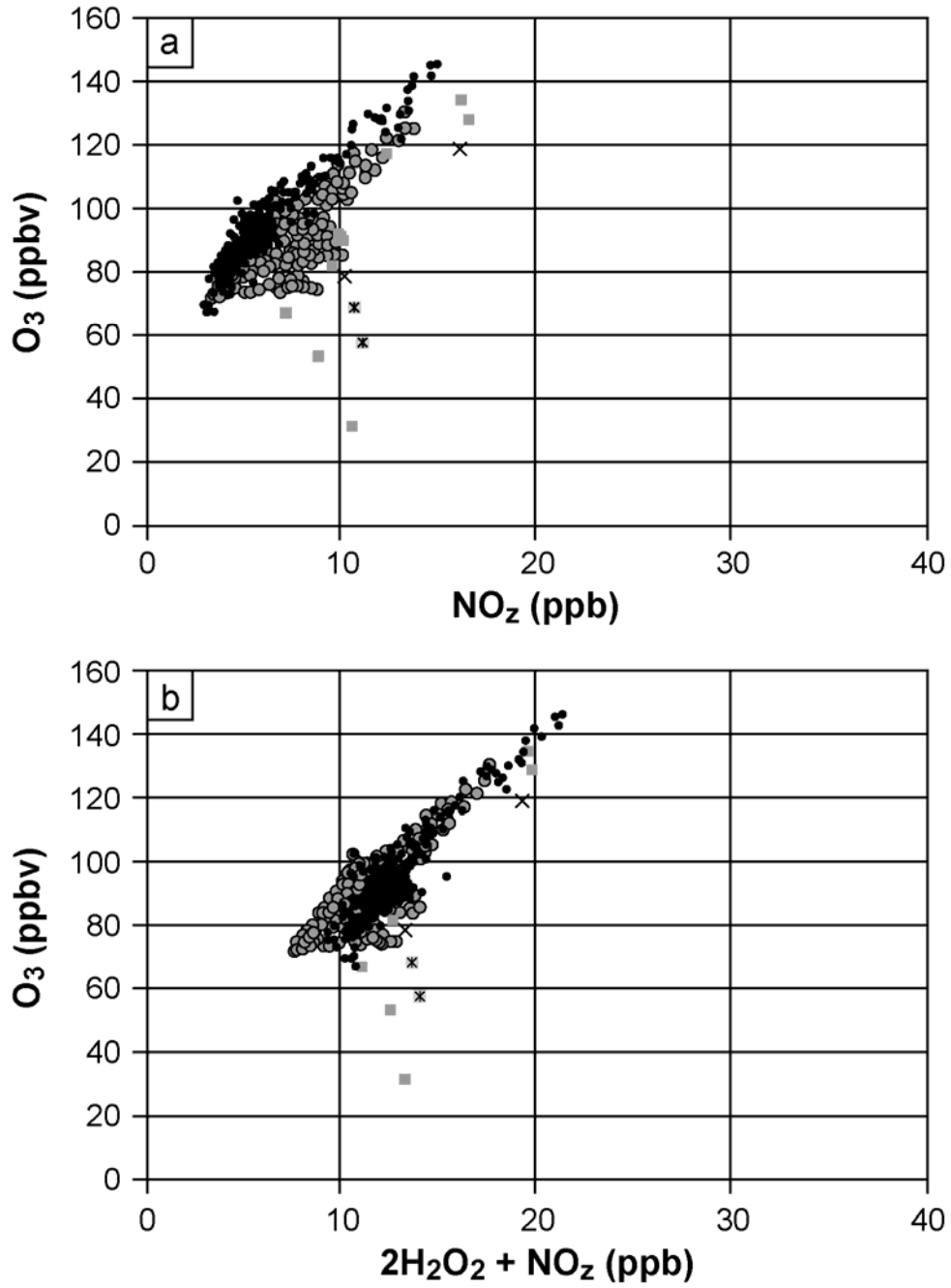


Figure AX2-20a,b. Evaluation of model versus: (a) measured O_3 versus NO_z and (b) O_3 versus the sum $2H_2O_2 + NO_z$ for Nashville, TN. The model values are classified as NO_x-limited (gray circles), NO_x-saturated (X's), mixed or near-zero sensitivity (squares), or dominated by NO_x titration (filled circles). Diamonds represent aircraft measurements.

Source: Sillman et al. (1998).

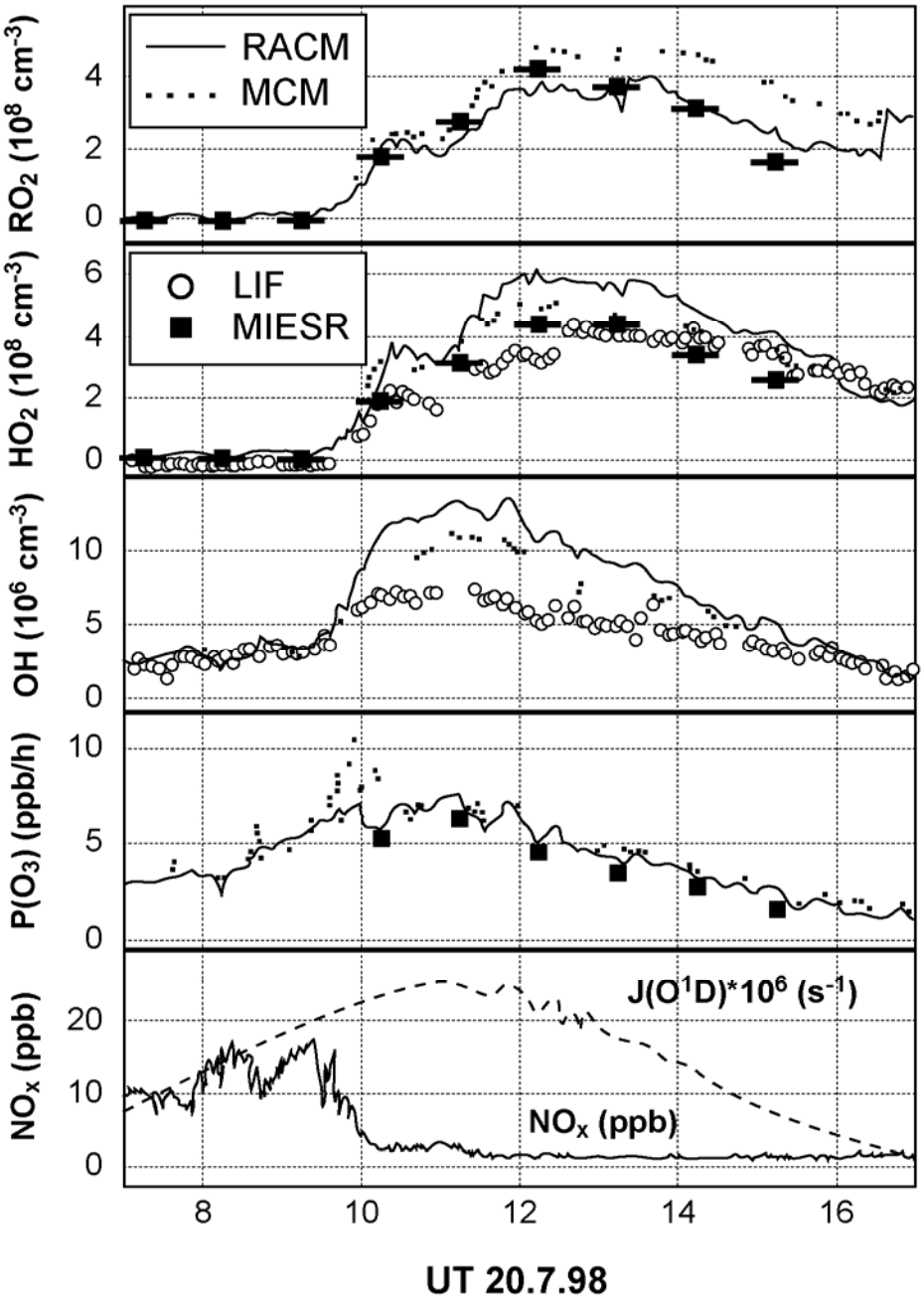


Figure AX2-21. Time series of concentrations of RO_2, HO_2, and OH radicals, local O_3 photochemical production rate and concentrations of NO_x from measurements made during BERLIOZ. Also shown are comparisons with results of photochemical box model calculations using the RACM and MCM chemical mechanisms.

Source: Volz-Thomas et al. (2003).

moderately overestimate HO_2 concentrations. Agreement between models and measurements is generally better for organic peroxy radicals, although the MCM appears to overestimate their concentrations somewhat. In general, the mechanisms reproduced the HO_2 to OH and RO_2 to OH ratios better than the individual measurements. The production of O_3 was found to increase linearly with NO (for NO <0.3 ppb) and to decrease with NO (for NO >0.5 ppb).

OH and HO_2 concentrations measured during the $PM_{2.5}$ Technology Assessment and Characterization Study conducted at Queens College in New York City in the summer of 2001 were also compared with those predicted by RACM (Ren et al., 2003). The ratio of observed to predicted HO_2 concentrations over a diurnal cycle was 1.24 and the ratio of observed to predicted OH concentrations was about 1.10 during the day, but the mechanism significantly underestimated OH concentrations during the night.

AX2.8 SAMPLING AND ANALYSIS OF NITROGEN AND SULFUR OXIDES

AX2.8.1 Availability and Accuracy of Ambient Measurements for NO_y

Section AX2.8.1-AX2.8.4 focus on current methods and on promising new technologies, but no attempt is made here to cover the extensive development of these methods or of methods such as wet chemical techniques, no longer in widespread use. More detailed discussions of these methods may be found elsewhere (U.S. Environmental Protection Agency, 1993, 1996). McClenny (2000), Parrish and Fehsenfeld (2000), and Clemitshaw (2004) reviewed methods for measuring NO_x and NO_y compounds. Discussions in Sections 2.8.1-2.8.4 center on chemiluminescence and optical Federal Reference and Equivalent Methods (FRM and FEM, respectively).

The use of methods such as observationally based methods or source apportionment models, either as stand-alone methods or as a basis for evaluating chemical transport models, is often limited by the availability and accuracy of measurements. Measured NO_x and speciated VOC concentrations are widely available in the United States through the PAMS network. However, challenges have been raised about both the accuracy of the measurements and their applicability.

The PAMs network currently includes measured NO and NO_x. However, Cardelino and Chameides (2000) reported that measured NO during the afternoon was frequently at or below the detection limit of the instruments (1 ppb), even in large metropolitan regions (Washington, DC; Houston, TX; New York, NY). Nitric **di**oxide measurements are made with commercial chemilluminescent detectors with hot molybdenum converters. However, these measurements typically include a wide variety of other reactive N species, such as organic nitrates in addition to NO_x, and cannot be interpreted as a "pure" NO_x measurement (see summary in Parrish and Fehsenfeld, 2000). Detection of these species can be considered an interference or a cross sensitivity useful for understanding the chemistry of the air.

Total reactive nitrogen (NO_y) is included in the PAMS network only at a few sites. The possible expansion of PAMS to include more widespread NO_y measurements has been suggested (McClenny, 2000). NO_y measurements are also planned for inclusion in the NCore network (U.S. EPA, 2005). A major issue to be considered when measuring NO_x and NO_y is the possibility that HNO_3, a major component of NO_y, is sometimes lost in inlet tubes and not measured (Luke et al., 1998; Parrish and Fehsenfeld, 2000). This problem is especially critical if measured NO_y is used to identify NO_x-limited versus NO_x-saturated conditions. The problem is substantially alleviated although not necessarily completely solved by using much shorter inlets on NO_y monitors than on NO_x monitors and by the use of surfaces less likely to take up HNO_3. The correlation between O_3 and NO_y differs for NO_x-limited versus NO_x-saturated locations, but this difference is driven primarily by differences in the ratio of O_3 to HNO_3. If HNO_3 were omitted from the NO_y measurements, then the measurements would represent a biased estimate and their use would be problematic.

AX2.8.1.1 Calibration Standards

Calibration gas standards of NO, in N_2 (certified at concentrations of approximately 5 to 40 ppm) are obtainable from the Standard Reference Material (SRM) Program of the National Institute of Standards and Technology (NIST), formerly the National Bureau of Standards (NBS), in Gaithersburg, MD. These SRMs are supplied as compressed gas mixtures at about 135 bar (1900 psi) in high-pressure aluminum cylinders containing 800 L of gas at standard temperature and pressure, dry (STPD) National Bureau of Standards, 1975; Guenther et al., 1996). Each cylinder is supplied with a certificate stating concentration and uncertainty. The concentrations are certified to be accurate to ±1 percent relative to the stated values. Because of

the resources required for their certification, SRMs are not intended for use as daily working standards, but rather as primary standards against which transfer standards can be calibrated.

Transfer stand-alone calibration gas standards of NO in N_2 (at the concentrations indicated above) are obtainable from specialty gas companies. Information as to whether a company supplies such mixtures is obtainable from the company, or from the SRM Program of NIST. These NIST Traceable Reference Materials (NTRMs) are purchased directly from industry and are supplied as compressed gas mixtures at approximately 135 bar (1900 psi) in high-pressure aluminum cylinders containing 4,000 L of gas at STPD. Each cylinder is supplied with a certificate stating concentration and uncertainty. The concentrations are certified to be accurate to within ±1 percent of the stated values (Guenther et al., 1996). Additional details can be found in the previous AQCD for O_3 (U.S. Environmental Protection Agency, 1996).

AX2.8.1.2 Measurement of Nitric Oxide

Gas-phase Chemiluminescence (CL) Methods

Nitric oxide can be measured reliably using the principle of gas-phase chemiluminescence induced by the reaction of NO with O_3 at low pressure. Modern commercial NO_x analyzers have sufficient sensitivity and specificity for adequate measurement in urban and many rural locations (U.S. Environmental Protection Agency, 1993, 1996, 2006). Research grade CL instruments have been compared under realistic field conditions to spectroscopic instruments, and the results indicate that both methods are reliable (at concentrations relevant to smog studies) to better than 15 percent with 95 percent confidence. Response times are on the order of 1 minute. For measurements meaningful for understanding O_3 formation, emissions modeling, and N deposition, special care must be taken to zero and calibrate the instrument frequently. A chemical zero, obtained by reacting the NO up-stream and out of view of the photomultiplier tube, is preferred because it accounts for interferences such as light emitting reactions with unsaturated hydrocarbons. Calibration should be performed with NTRM-of compressed NO in N_2. Standard additions of NO at the inlet will account for NO loss or conversion to NO_2 in the lines. In summary, CL methods, when operated carefully in an appropriate manner, can be suitable for measuring or monitoring NO (e.g., Crosley, 1996).

Spectroscopic Methods for Nitric Oxide

Nitric oxide has also been successfully measured in ambient air with direct spectroscopic methods; these include two-photon laser-induced fluorescence (TPLIF), tunable diode laser absorption spectroscopy (TDLAS), and two-tone frequency-modulated spectroscopy (TTFMS). These were reviewed thoroughly in the previous AQCD and will be only briefly summarized here. The spectroscopic methods demonstrate excellent sensitivity and selectivity for NO with detection limits on the order of 10 ppt for integration times of 1 min. Spectroscopic methods compare well with the CL method for NO in controlled laboratory air, ambient air, and heavily polluted air (e.g., Walega et al., 1984; Gregory et al., 1990; Kireev et al., 1999). These spectroscopic methods remain in the research arena due to their complexity, size, and cost, but are essential for demonstrating that CL methods are reliable for monitoring NO concentrations involved in O_3 formation—from around 20 ppt to several hundred of ppb.

Atmospheric pressure laser ionization followed by mass spectroscopy has also been deployed for detection of NO and NO_2. Garnica et al. (2000) describe a technique involving selective excitation at one wavelength followed by ionization at a second wavelength. They report good selectivity and detection limits well below 1 ppb. The practicality of the instrument for ambient monitoring, however, has yet to be demonstrated.

AX2.8.1.3 Measurements of Nitrogen Dioxide

Gas-Phase Chemiluminescence Methods

Reduction of NO_2 to NO, on the surface of a heated (to 300 to 400 °C) molybdenum oxide substrate followed by detection of the chemiluminescence produced during the reaction of NO with O_3 at low pressure as described earlier for measurement of NO serves as the basis of the FRM for measurement of ambient NO_2. However, the substrate used in the reduction of NO_2 to NO is not specific to NO_2; hence the chemiluminescence analyzers are subject to interference nitrogen oxides other than NO_2 produced by oxidized NO_y compounds, or NO_z. Thus, this technique will overestimate NO_2 concentrations particularly in areas downwind of sources of NO and NO_2 as NO_x is oxidized to NO_z in the form of PANs and other organic nitrates, and HNO_3 and HNO_4. Many of these compounds are reduced at the catalyst with nearly the same efficiency as NO_2. Interferences have also been found from a wide range of other compounds as described in the latest AQCD for NO_2.

Other Methods

Nitrogen dioxide can be selectively converted to NO by photolysis. For example, (Ryerson et al., 2000) developed a gas-phase chemiluminescence method using a photolytic converter based on a Hg lamp with increased radiant intensity in the region of peak NO_2 photolysis (350 to 400 nm) and producing conversion efficiencies of 70% or more in less than 1 s. Metal halide lamps with conversion efficiency of about 50% and accuracy on the order of 20% (Nakamura, et al., 2003) have been used. Because the converter produces little radiation at wavelengths less than 350 nm, interferences from HNO_3 and PAN are minimal. Alternative methods to photolytic reduction followed by CL are desirable to test the reliability of this widely used technique. Any method based on a conversion to measured species presents potential for interference a problem. Several atmospheric species, PAN and HO_2NO_2 for example, dissociate to NO_2 at higher temperatures.

Laser induced fluorescence for NO_2 detection involves excitation of atmospheric NO_2 with laser light emitted at wavelengths too long to induce photolysis. The resulting excited molecules relax in a photoemissive mode and the fluorescing photons are counted. Because collisions would rapidly quench the electronically excited NO_2, the reactions are conducted at low pressure. Matsumi et al. (2001) describe a comparison of LIF with a photofragmentation chemiluminescence instrument. The LIF system involves excitation at 440 nm with a multiple laser system. They report sensitivity of 30 ppt in 10 s and good agreement between the two methods under laboratory conditions at mixing ratios up to 1.0 ppb. This high-sensitivity LIF system has yet to undergo long-term field tests. Cleary et al. (2002) describe field tests of a system that uses continuous, supersonic expansion followed by excitation at 640 nm with a commercial cw external-cavity tunable diode laser. More recently, LIF has been successfully used to detect NO_2 with accuracy of about 15% and detections limits well below 1 ppb. When coupled with thermal dissociation, the technique also measures peroxy nitrates such as PAN, alkyl nitrates, HNO_4 and HNO_3 (Cohen, 1999; Day et al., 2002; Farmer et al., 2006; Perez et al., 2007; Thornton et al., 2003). This instrument can have very fast sampling rates be fast (>1 Hz) and shows good correlation with chemiluminescent techniques, but remains a research-grade device.

Nitrogen Dioxide can be detected by differential optical absorption spectroscopy (DOAS) in an open, long-path system by measuring narrow band absorption features over a background

of broad band extinction (e.g., Stutz et al., 2000; Kim and Kim, 2001). A DOAS system manufactured by OPSIS is designated as a Federal Equivalent Method for measuring NO_2. DOAS systems can also be configured to measure NO, HONO, and NO_3 radicals. Typical detection limits are 0.2 to 0.3 ppbv for NO, 0.05 to 0.1 ppbv for NO_2, 0.05 to 0.1 ppbv for HONO, and 0.001 to 0.002 ppbv for NO_3, at path lengths of 0.2, 5, 5, and 10 km, respectively. The obvious advantage compared to fixed point measurements is that concentrations relevant to a much larger area are obtained, especially if multiple targets are used. At the same time, any microenvironmental artifacts are minimized over the long path integration. A major limitation in this technique had involved inadequate knowledge of absorption cross sections. Harder et al. (1997) conducted an experiment in rural Colorado involving simultaneous measurements of NO_2 by DOAS and by photolysis followed by chemiluminescence. They found differences of as much as 110% in clean air from the west, but for NO_2 mixing ratios in excess of 300 ppt, the two methods agreed to better than 10%. Stutz (2000) cites two intercomparisons of note. Nitric oxide was measured by DOAS, by photolysis of NO_2 followed by chemiluminescence, and by LIF during July 1999 as part of the SOS in Nashville, TN. On average, the three methods agreed to within 2%, with some larger differences likely caused by spatial variability over the DOAS path. In another study in Europe, and a multi-reflection set-up over a 15 km path, negated the problem of spatial averaging here agreement with the chemiluminescence detector following photolytic conversion was excellent (slope = 1.006 ± 0.005; intercept = 0.036 ± 0.019; r = 0.99) over a concentration range from about 0.2 to 20 ppbv.

Nitric oxide can also be detected from space with DOAS-like UV spectroscopy techniques (Kim et al., 2006; Ma et al., 2006). These measurements appear to track well with emissions estimates and can be a useful indicator of column content as well as for identifying hot spots in sources. **See also** Richter, et al., 2005. Leigh (2006) report on a DOAS method that uses the sun as a light source and compares well with an in situ chemiluminescence detector in an urban environment.

Chemiluminescence on the surface of liquid Luminol has also been used for measurement of NO_2 (Gaffney et al., 1998; Kelly et al., 1990; Marley et al., 2004; Nikitas et al., 1997; Wendel et al., 1983). This technique is sensitive and linear, and more specific than hot MoOx. Luminol does not emit light when exposed to NHO_3 or alkyl nitrates, but does react with PAN. This interference can be removed by chromatographic separation prior to detection and the resulting

measurement compares well with more specific techniques for moderate to high (≥1 ppb) mixing ratios of NO_2.

Several tunable diode laser spectroscopy techniques have been used successfully for NO_2 detection (Eisele et al., 2003; Osthoff et al., 2006). These devices remain research grade instruments, not yet practical for urban monitoring.

Measurements of Total Oxidized Nitrogen Species, NO_y

Gold catalyzed CO, or H_2 reduction or as conversion on hot molybdenum oxide catalyst have been used to reduce NO_y to NO before then detection by chemiluminescence (Fehsenfeld et al., 1987; Crosley, 1996). Both techniques offer generally reliable measurements, with response times on the order of 60 s and a linear dynamic range demonstrated in field intercomparisons from about 10 ppt to 10's of ppb. Under certain conditions, HCN, NH_3, RNO_2, and CH_3CN can be converted to NO, but at normal concentrations and humidity these are minor interferences. Thermal decomposition followed by LIF has also been used for NO_y detection, as described above. In field comparisons, instruments based on these two principles generally showed good agreement (Day et al., 2002). The experimental uncertainty is estimated to be of 15-30%.

AX2.8.1.4 Monitoring for NO_2 Compliance Versus Monitoring for Ozone Formation

Regulatory measurements of NO_2 have been focused on demonstrating compliance with the NAAQS for NO_2. Today, few locations violate that standard, but NO_2 and related NO_y compounds remain among the most important atmospheric trace gases to measure and understand. Commercial instruments for NO/NO_x detection are generally constructed with an internal converter for reduction of NO_2 to NO, and generate a signal referred to as NO_x. These converters, generally constructed of molybdenum oxides (MoOx), reduce not only NO_2 but also most other NO_y species. Unfortunately, with an internal converter, the instruments may not give a faithful indication of NO_y either—reactive species such as HNO_3 will adhere to the walls of the inlet system. Most recently, commercial vendors such as Thermo Environmental (Franklin, MA) have offered NO/NO_y detectors with external Mo converters. If such instruments are calibrated through the inlet with a reactive nitrogen species such as propyl nitrate, they give accurate measurements of total NO_y, suitable for evaluation of photochemical models. (Crosley, 1996; Fehsenfeld et al., 1987; Nunnermacker et al., 1998; Rodgers and Davis, 1989). Under conditions

of fresh emissions, such as in urban areas during the rush hour, $NO_y \approx NO_x$ and these monitors can be used for testing emissions inventories (Dickerson, et al., 1995; Parrish, 2006). The State of Maryland for example is making these true NO_y measurements at the Piney Run site in the western part of the state. These data produced at this site can be more reliably compared to the output of CMAQ and other chemical transport models.

Summary of Methods for Measuring NO_2

A variety of techniques exist for reliable monitoring of atmospheric NO_2 and related reactive nitrogen species. For demonstration of compliance with the NAAQS for NO_2, commercial chemiluminescence instruments are adequate. For certain conditions, luminol chemiluminescence is adequate. Precise measurements of NO_2 can be made with research grade instruments such as LIF and TDLS. For path-integrated concentration determinations UV spectroscopic methods provide useful information. Commercial NO_x instruments are sensitive to other NO_y species, but do not measure NO_y quantitatatively. NO_y instruments with external converters offer measurements more useful for comparison to chemical transport model calculations.

AX2.8.2 Measurements of HNO_3

Accurate measurement of HNO_3, has presented a long-standing analytical challenge to the atmospheric chemistry community. In this context, it is useful to consider the major factors that control HNO_3 partitioning between the gas and deliquesced-particulate phases in ambient air. In equation form,

$$HNO_{3g} \overset{K_H}{\longleftrightarrow} [HNO_{3aq}] \overset{K_a}{\longleftrightarrow} [H^+] + [NO_3^-] \qquad \text{(AX2-51)}$$

where K_H is the Henry's Law constant in M atm^{-1} and K_a is the acid dissociation constant in M.

Thus, the primary controls on HNO_3 phase partitioning are its thermodynamic properties (K_H, K_a, and associated temperature corrections), aerosol liquid water content (LWC), solution pH, and kinetics. Aerosol LWC and pH are controlled by the relative mix of different acids and bases in the system, hygroscopic properties of condensed compounds, and meteorological conditions (RH, temperature, and pressure). It is evident from relationship AX2-51 that, in the presence of chemically distinct aerosols of varying acidities (e.g., super-µm predominantly sea

salt and sub-µm predominantly S aerosol), HNO_3 will partition preferentially with the less-acidic particles; and this is consistent with observations (e.g., Huebert et al., 1996; Keene and Savoie, 1998; Keene et al., 2002). Kinetics are controlled by atmospheric concentrations of HNO_3 vapor and particulate NO_3^- and the size distribution and corresponding atmospheric lifetimes of particles against deposition. Sub-µm diameter aerosols typically equilibrate with the gas phase in seconds to minutes while super-um aerosols require hours to a day or more (e.g., Meng and Seinfeld, 1996; Erickson et al., 1999). Consequently, smaller aerosol size fractions are typically close to thermodynamic equilibrium with respect to HNO_3 whereas larger size fractions (for which atmospheric lifetimes against deposition range from hours to a few days) are often undersaturated (e.g., Erickson et al., 1999; Keene and Savioe, 1998).

Many sampling techniques for HNO_3 (e.g., annular denuder, standard filterpack and mist-chamber samplers) employ upstream prefilters to remove particulate species from sample air. However, when chemically distinct aerosols with different pHs (e.g., sea salt and S aerosols) mix together on a bulk filter, the acidity of the bulk mixture will be greater than that of the less acidic aerosols with which most NO_3^- is associated. This change in pH may cause the bulk mix to be supersaturated with respect to HNO_3 leading to volatilization and, thus, positive measurement bias in HNO_3 sampled downstream. Alternatively, when undersaturated super-µm size fractions (e.g., sea salt) accumulate on a bulk filter and chemically interact over time with HNO_3 in the sample air stream, scavenging may lead to negative bias in HNO_3 sampled downstream. Because the magnitude of both effects will vary as functions of the overall composition and thermodynamic state of the multiphase system, the combined influence can cause net positive or net negative measurement bias in resulting data. Pressure drops across particle filters can also lead to artifact volatilization and associated positive bias in HNO_3 measured downstream.

Widely used methods for measuring HNO_3 include standard filterpacks configured with nylon or alkaline-impregnated filters (e.g., Goldan et al., 1983; Bardwell et al., 1990), annular denuders (EPA Method IP-9), and standard mist chambers (Talbot et al., 1990). Samples from these instruments are typically analyzed by ion chromatography. Intercomparisons of these measurement techniques (e.g., Hering et al., 1988; Tanner et al., 1989; Talbot et al., 1990) report differences on the order of a factor of two or more.

More recently, sensitive HNO_3 measurements based on the principle of Chemical Ionization Mass Spectroscopy (CIMS) have been reported (e.g., Huey et al., 1998; Mauldin

et al., 1998; Furutani and Akimoto, 2002; Neuman et al., 2002). CIMS relies on selective formation of ions such as $SiF_5^- \cdot HNO_3$ or $HSO_4^- \cdot HNO_3$ followed by detection via mass spectroscopy. Two CIMS techniques and a filter pack technique were intercompared in Boulder, CO (Fehsenfeld et al., 1998). Results indicated agreement to within 15% between the two CIMS instruments and between the CIMS and filterpack methods under relatively clean conditions with HNO_3 mixing ratios between 50 and 400 pptv. In more polluted air, the filterpack technique generally yielded higher values than the CIMS suggesting that interactions between chemically distinct particles on bulk filters is a more important source of bias in polluted continental air. Differences were also greater at lower temperature when particulate NO_3^- corresponded to relatively greater fractions of total NO_3^-.

AX2.8.3 Techniques for Measuring Other NO_y Species

Methods for sampling and analysis of alkyl nitrates in the atmosphere have been reviewed by Parrish and Fehsenfeld (2000). Peroxyacetylnitrate, PPN, and MPAN are typically measured using a chromatograph followed by electron capture detectors or GC/ECD (e.g., Gaffney et al., 1998), although other techniques such as FTIR could also be used. Field measurements are made using GC/ECD with a total uncertainty of ± 5 pptv + 15% (Roberts et al., 1998).

In the IMPROVE network and in the EPA's speciation network, particulate nitrate in the $PM_{2.5}$ size range is typically collected on nylon filters downstream of annular denuders coated with a basic solution capable of removing acidic gases such as HNO_3, HNO_2, and SO_2. Filter extracts are then analyzed by ion chromatography (IC) for nitrate, sulfate, and chloride. Nitrite ions are also measured by this technique but their concentrations are almost always beneath detection limits. However, both of these networks measure nitrate only in the $PM_{2.5}$ fraction. Because of interactions with more highly acidic components on filter surfaces, there could be volatilization of nitrate in PM_{10} samples. These effects are minimized if separate aerosol size fractions are collected, i.e., the more acidic $PM_{2.5}$ and the more alkaline $PM_{10\text{-}2.5}$ as in a dichotomous sampler or multistage impactor.

AX2.8.4 Remote Sensing of Tropospheric NO_2 Columns for Surface NO_x Emissions and Surface NO_2 Concentrations

Table AX2-3 contains an overview of the three satellite instruments that are used retrieve tropospheric NO_2 columns from measurements of solar backscatter. All three instruments are in polar sun-synchronous orbits with global measurements in the late morning and early afternoon. The spatial resolution of the measurement from SCIAMACHY is 7 times better than that from GOME (Ozone Monitoring Instrument), and that from OMI (Ozone Monitoring Instrument) is 40 times better than that from GOME.

Figure AX2-22 shows tropospheric NO_2 columns retrieved from SCIAMACHY. Pronounced enhancements are evident over major urban and industrial emissions. The high degree of spatial heterogeneity over the southwestern United States provides empirical evidence that most of the tropospheric NO_2 column is concentrated in the lower troposphere. Tropospheric NO_2 columns are more sensitive to NO_x in the lower troposphere than in the upper troposphere (Martin et al., 2002). This sensitivity to NO_x in the lower troposphere is due to the factor of 25 decrease in the NO_2/NO ratio from the surface to the upper troposphere (Bradshaw et al., 1999) that is driven by the temperature dependence of the NO + O_3 reaction. Martin et al. (2004a) integrated in situ airborne measurements of NO_2 and found that during summer the lower mixed layer contains 75% of the tropospheric NO_2 column over Houston and Nashville. However, it should be noted that these measurements are also sensitive to surface albedo and aerosol loading.

The retrieval involves three steps: (1) determining total NO_2 line-of-sight (slant) columns by spectral fitting of solar backscatter measurements, (2) removing the stratospheric columns by using data from remote regions where the tropospheric contribution to the column is small, and (3) applying an air mass factor (AMF) for the scattering atmosphere to convert tropospheric slant columns into vertical columns. The retrieval uncertainty is determined by (1) and (2) over remote regions where there is little tropospheric NO_2, and by (3) over regions in regions of elevated tropospheric NO_2 (Martin et al., 2002; Boersma et al., 2004).

The paucity of in situ NO_2 measurements motivates the inference of surface NO_2 concentrations from satellite measurements of tropospheric NO_2 columns. This prospect would take advantage of the greater sensitivity of tropospheric NO_2 columns to NO_x in the lower

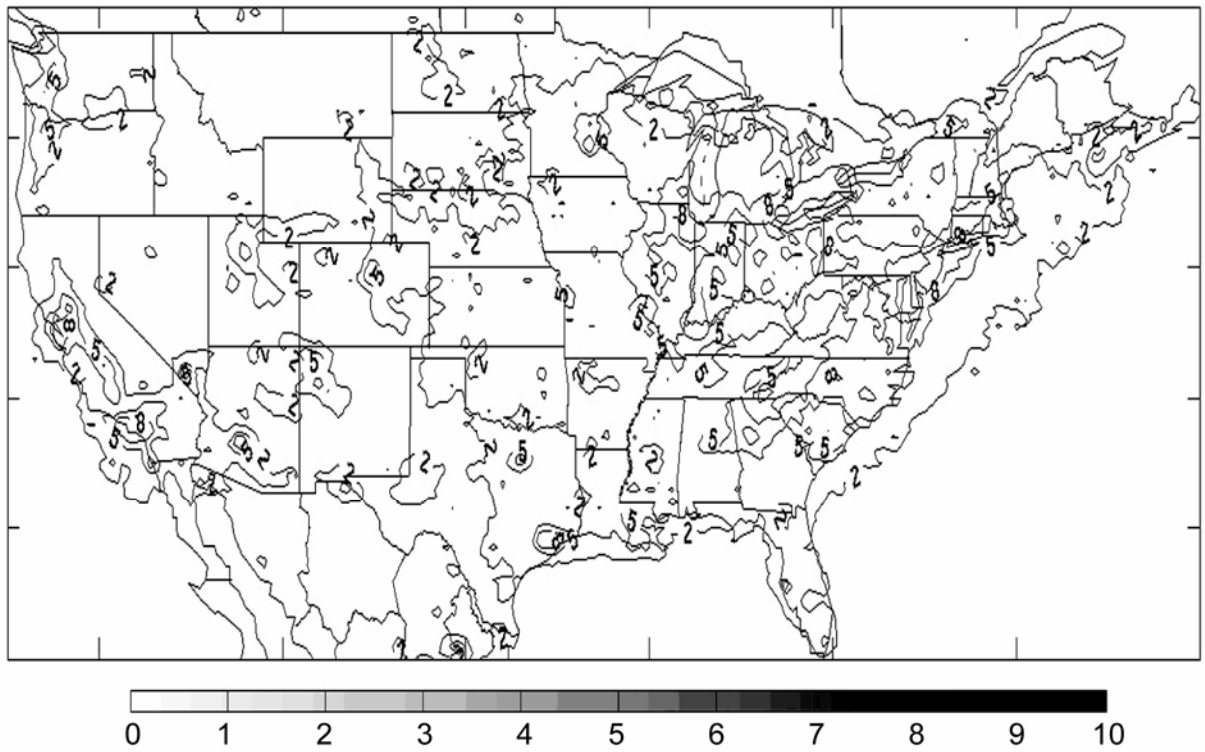


Figure AX2-22. Tropospheric NO_2 columns (molecules NO_2/ cm^2) retrieved from the SCIAMACHY satellite instrument for 2004-2005.

Source: Martin et al. (2006).

troposphere than in the upper troposphere as discussed earlier. Tropospheric NO_2 columns show a strong correlation with in situ NO_2 measurements in northern Italy (Ordonez et al., 2006).

Quantitative calculation of surface NO_2 concentrations from a tropospheric NO_2 column would require information on the relative vertical profile. Comparison of vertical profiles of NO_2 in a chemical transport model (GEOS-Chem) versus in situ measurements over and downwind of North America shows a high degree of consistency (Martin et al., 2004b; Martin et al., 2006), suggesting that chemical transport models could be used to infer the relationship between surface NO_2 concentrations and satellite observations of the tropospheric NO_2 column.

However, the satellites carrying the spectrometer (GOME/SCIAMACHY/OMI) are in near polar, sun-synchronous orbits. As a result, these measurements are made only once per day, typically between about 10:00 to 11:00 a.m. or **1 p.m.** local time, during a brief overflight. Thus the utility of these measurements is limited as they would likely miss short-term features.

AX2.8.5 SAMPLING AND ANALYSIS FOR SO_2

Currently, ambient SO_2 is measured using instruments based on pulsed fluorescence. The UV fluorescence monitoring method for atmospheric SO_2 was developed to improve upon the flame photometric detection (FPD) method for SO_2, which in turn had displaced the pararosaniline wet chemical method for SO_2 measurement. The pararosaniline method is still the FRM for atmospheric SO_2, but is rarely used because of its complexity and slow response, even in its automated forms. Both the UV fluorescence and FPD methods are designated as FEMs by the EPA, but UV fluorescence has largely supplanted the FPD approach because of the UV method's inherent linearity, sensitivity, and the absence of consumables, such as the hydrogen gas needed for the FPD method.

Basically, SO_2 molecules absorb ultraviolet (UV) light at one wavelength and emit UV light at longer wavelengths. This process is known as fluorescence, and involves the excitation of the SO_2 molecule to a higher energy (singlet) electronic state. Once excited, the molecule decays non-radiatively to a lower energy electronic state from which it then decays to the original, or ground, electronic state by emitting a photon of light at a longer wavelength (i.e., lower energy) than the original, incident photon. The process can be summarized by the following equations:

$$SO_2 + h\nu_1 \rightarrow SO_2^* \qquad \text{(AX2-52)}$$

$$SO_2^* \rightarrow SO_2 + h\nu_2 \qquad \text{(AX2-53)}$$

where SO_2^* represents the excited state of SO_2, $h\nu_1$, and $h\nu_2$ represent the energy of the excitation and fluorescence photons, respectively, and $h\nu_2 < h\nu_1$. The intensity of the emitted light is proportional to the number of SO_2 molecules in the sample gas.

In commercial analyzers, light from a high intensity UV lamp passes through a bandwidth filter, allowing only photons with wavelengths around the SO_2 absorption peak (near 214 nm) to enter the optical chamber. The light passing through the source bandwidth filter is collimated using a UV lens and passes through the optical chamber, where it is detected on the opposite side of the chamber by the reference detector. A photomultiplier tube (PMT) is offset from and placed perpendicular to the light path to detect the SO_2 fluorescence. Since the SO_2 fluorescence (330 nm) is at a wavelength that is different from the excitation wavelength, an

optical bandwidth filter is placed in front of the PMT to filter out any stray light from the UV lamp. A lens is located between the filter and the PMT to focus the fluorescence onto the active area of the detector and optimize the fluorescence signal. The Detection Limit (DL) for a non-trace level SO_2 analyzer is 10 parts per billion (ppb) (Code of Federal Regulations, Volume 40, Part 53.23c). The SO_2 measurement method is subject to both positive and negative interference.

Sources of Positive Interference

The most common source of interference is from other gases that fluoresce in a similar fashion to SO_2 when exposed to far UV radiation. The most significant of these are polycyclic aromatic hydrocarbons (PAHs); of which naphthalene is a prominent example. Xylene is another hydrocarbon that can cause interference.

Such compounds absorb UV photons and fluoresce in the region of the SO_2 fluorescence. Consequently, any such aromatic hydrocarbons that are in the optical chamber can act as a positive interference. To remove this source of interference, the high sensitivity SO_2 analyzers, such as those to be used in the NCore network (U.S. Environmental Protection Agency, 2005), have hydrocarbon scrubbers to remove these compounds from the sample stream before the sample air enters the optical chamber.

Another potential source of positive interference is nitric oxide (NO). NO fluoresces in a spectral region that is close to the SO_2 fluorescence. However, in high sensitivity SO_2 analyzers, the bandpass filter in front of the PMT is designed to prevent NO fluorescence from reaching the PMT and being detected. Care must be exercised when using multicomponent calibration gases containing both NO and SO_2 that the NO rejection ratio of the SO_2 analyzer is sufficient to prevent NO interference. The most common source of positive bias (as constrasted with positive spectral interference) in high-sensitivity SO_2 monitoring is stray light reaching the optical chamber. Since SO_2 can be electronically excited by a broad range of UV wavelengths, any stray light with an appropriate wavelength that enters the optical chamber can excite SO_2 in the sample and increase the fluorescence signal.

Furthermore, stray light at the wavelength of the SO_2 fluorescence that enters the optical chamber may impinge on the PMT and increase the fluorescence signal. Several design features are incorporated to minimize the stray light that enters the chamber. These features include the use of light filters, dark surfaces, and opaque tubing to prevent light from entering the chamber.

Luke (1997) reported the positive artifacts of a modified pulsed fluorescence detector generated by the co-existence of NO, CS_2, and a number of highly fluorescent aromatic hydrocarbons such as benzene, toluene, o-xylene, m-xylene, p-xylene, m-ethyltoluene, ethylbenzene, and 1,2,4-trimethylbenzene. The positive artifacts could be reduced by using a hydrocarbon "kicker" membrane. At a flow rate of 300 standard cc min^{-1} and a pressure drop of 645 torr across the kicker, the interference from ppm levels of many aromatic hydrocarbons was eliminated entirely.

Nicks and Benner (2001) described a sensitive SO_2 chemiluminescence detector, which was based on a differential measurement where response from ambient SO_2 is determined by the difference between air containing SO_2 and air scrubbed of SO_2 where both air samples contain other detectable sulfur species, and the positive artifact could also be reduced through this way.

Sources of Negative Interference

Nonradiative deactivation (quenching) of excited SO_2 molecules can occur from collisions with common molecules in air, including nitrogen, oxygen, and water. During collisional quenching, the excited SO_2 molecule transfers energy, kinetically allowing the SO_2 molecule to return to the original lower energy state without emitting a photon. Collisional quenching results in a decrease in the SO_2 fluorescence and results in the underestimation of SO_2 concentration in the air sample. The concentrations of nitrogen and oxygen are constant in the ambient air, so quenching from those species at a surface site is also constant, but the water vapor content of air can vary. Luke (1997) reported that the response of the detector could be reduced by about 7% and 15% at water vapor mixing ratios of 1 and 1.5 mole percent (RH = 35 to 50% at 20-25 °C and 1 atm for a modified pulsed fluorescence detector (Thermo Environmental Instruments, Model 43s). Condensation of water vapor in sampling lines must be avoided, as it can absorb SO_2 from the sample air. The simplest approach to avoid condensation is to heat sampling lines to a temperature above the expected dew point, and within a few degrees of the controlled optical bench temperature. At very high SO_2 concentrations, reactions between electronically excited SO_2 and ground state SO_2 to form SO_3 and SO might occur (Calvert et al., 1978). However, this possibility has not been examined.

Other Techniques for Measuring SO_2

A more sensitive SO_2 measurement method than the UV-fluorescence method was reported by Thornton et al (2002). Thornton et al (2002) reported an atmospheric pressure ionization mass spectrometer. The high measurement precision and instrument sensitivity were achieved by adding isotopically labeled SO_2 ($^{34}S^{16}O_2$) continuously to the manifold as an internal standard. Field studies showed that the method precision was better than 10% and the limit of detection was less than 1 pptv for a sampling interval of 1s.

Sulfur Dioxide can be measured by LIF at around 220 nm (Matsumi et al. (2005). Because the laser wavelength is alternately tuned to an SO_2 absorption peak at 220.6 and bottom at 220.2 nm, and the difference signal at the two wavelengths is used to extract the SO_2 concentration, the technique eliminates interference from either absorption or fluorescence by other species and has high sensitivity (5 pptv in 60 sec). Sulfur Dioxide can also be measured by the same DOAS instrument that can measure NO_2.

Photoacoutsic techniques have been employed for SO_2 detection, but they generally have detection limits suitable only for source monitoring (Gondal, 1997; Gondal and Mastromarino, 2001).

Chemical Ionization Mass Spectroscopy (CIMS) utilizes ionization via chemical reactions in the gas phase to determine an unknown sample's mass spectrum and identity. High sensitivity (10 ppt or better) has been achieved with uncertainty of ~15% when a charcoal scrubber is used for zeroing and the sensitivity is measured with isotopically labeled $^{34}SO_2$ (Hanke et al., 2003; Huey et al., 2004; Hennigan et al., 2006).

AX2.8.6 Sampling and Analysis for Sulfate, Nitrate, and Ammonium

Sampling Artifacts

Sulfate, nitrate, and ammonium are commonly present in $PM_{2.5}$. Most $PM_{2.5}$ samplers **have** a size-separation device to separate particles so that only those particles approximately 2.5 μm or less are collected on the sample filter. Air is drawn through the sample filter at a controlled flow rate by a pump located downstream of the sample filter. The systems have two critical flow rate components for the capture of fine particulate: (1) the flow of air through the sampler must be at a flow rate that ensures that the size cut at 2.5 μm occurs; and (2) the flow

rate must be optimized to capture the desired amount of particulate loading with respect to the analytical method detection limits.

When using the system described above to collect sulfate, nitrate and particulate ammonium, sampling artifacts can occur because of: (1) positive sampling artifact for sulfate, nitrate, and particulate ammonium due to chemical reaction; and (2) negative sampling artifact for nitrate and ammonium due to the decomposition and evaporation.

Sampling and Analysis Techniques

Denuder-Filter Based Sampling and Analysis Techniques for Sulfate, Nitrate, and Ammonium

There are two major PM speciation ambient air-monitoring networks in the U.S.: the Speciation Trend Network (STN), and the Interagency Monitoring of Protected Visual Environments (IMPROVE) network. The current STN samplers include three filters: (1) Teflon for equilibrated mass and elemental analysis including elemental sulfur; (2) a HNO_3 denuded nylon filter for ion analysis including NO_3 and SO_4, (3) a quartz-fiber filter for elemental and organic carbon. The IMPROVE sampler, which collects two 24-h samples per week, simultaneously collects one sample of PM_{10} on a Teflon filter, and three samples of $PM_{2.5}$ on Teflon, nylon, and quartz filters. $PM_{2.5}$ mass concentrations are determined gravimetrically from the $PM_{2.5}$ Teflon filter sample. The $PM_{2.5}$ Teflon filter sample is also used to determine concentrations of selected elements. The $PM_{2.5}$ nylon filter sample, which is preceded by a denuder to remove acidic gases, is analyzed to determine nitrate and sulfate aerosol concentrations. Finally, the $PM_{2.5}$ quartz filter sample is analyzed for OC and EC using the thermal-optical reflectance (TOR) method. The STN and the IMPROVE networks represent a major advance in the measurement of nitrate, because the combination of a denuder (coated with either Na_2CO_3 or MgO) to remove HNO_3 vapor and a Nylon filter to adsorb HNO_3 vapor volatilizing from the collected ammonium nitrate particles overcomes the loss of nitrate from Teflon filters.

The extent to which sampling artifacts for particulate NH_3^+ have been adequately addressed in the current networks is not clear. Recently, new denuder-filter sampling systems have been developed to measure sulfate, nitrate, and ammonium with an adequate correction of ammonium sampling artifacts. The denuder-filter system, Chembcomb Model 3500 speciation sampling cartridge developed by Rupprecht & Patashnick Co, Inc. could be used to collect

nitrate, sulfate, and ammonium simultaneously. The sampling system contains a single-nozzle size-selective inlet, two honeycomb denuders, the aerosol filter and two backup filters (Keck and Wittmaack, 2005). The first denuder in the system is coated with 0.5% sodium carbonate and 1% glycerol and collects acid gases such as HCL, SO_2, HONO, and HNO_3. The second denuder is coated with 0.5% phosphoric acid in methanol for collecting NH_3. Backup filters collect the gases behind denuded filters. The backup filters are coated with the same solutions as the denuders. A similar system based on the same principle was applied by Possanzini et al. (1999). The system contains two NaCl-coated annular denuders followed by other two denuders coated with $NaCO_3$/glycerol and citric acid, respectively. This configuration was adopted to remove HNO_3 quantitatively on the first NaCl denuder. The third and forth denuder remove SO_2 and NH_3, respectively. A polyethylene cyclone and a two-stage filter holder containing three filters is placed downstream of the denuders. Aerosol fine particles are collected on a Teflon membrane. A backup nylon filter and a subsequent citric acid impregnated filter paper collect dissociation products (HNO_3 and NH_3) of ammonium nitrate evaporated from the filtered particulate matter.

Several traditional and new methods could be used to quantify elemental S collected on filters: energy dispersive X-ray fluorescence, synchrotron induced X-ray fluorescence, proton induced X-ray emission (PIXE), total reflection X-ray fluorescence, and scanning electron microscopy. Energy dispersive X-ray fluorescence (EDXRF) (Method IO-3.3, U.S. EPA, 1997; see 2004 PM CD for details) and PIXE are the most commonly used methods. Since sample filters often contain very small amounts of particle deposits, preference is given to methods that can accommodate small sample sizes and require little or no sample preparation or operator time after the samples are placed into the analyzer. X-ray fluorescence (XRF) meets these needs and leaves the sample intact after analysis so it can be submitted for additional examinations by other methods as needed. To obtain the greatest efficiency and sensitivity, XRF typically places the filters in a vacuum which may cause volatile compounds (nitrates and organics) to evaporate. As a result, species that can volatilize such as ammonium nitrate and certain organic compounds can be lost during the analysis. The effects of this volatilization are important if the PTFE filter is to be subjected to subsequent analyses of volatile species.

Polyatomic ions such as sulfate, nitrate, and ammonium are quantified by methods such as ion chromatography (IC) (an alternative method commonly used for ammonium analysis is

automated colorimetry). All ion analysis methods require a fraction of the filter to be extracted in deionized distilled water for sulfate and $NaCO_3$/$NaHCO_3$ solution for nitrate and then filtered to remove insoluble residues prior to analysis. The extraction volume should be as small as possible to avoid over-diluting the solution and inhibiting the detection of the desired constituents at levels typical of those found in ambient $PM_{2.5}$ samples. During analysis, the sample extract passes through an ion-exchange column which separates the ions in time for individual quantification, usually by an electroconductivity detector. The ions are identified by their elution/retention times and are quantified by the conductivity peak area or peak height.

In a side-by-side comparison of two of the major aerosol monitoring techniques (Hains et al., 2007), $PM_{2.5}$ mass and major contributing species were well correlated among the different methods with r-values in excess of 0.8. Agreement for mass, sulfate, OC, TC, and ammonium was good while that for nitrate and BC was weaker. Based on reported uncertainties, however, even daily concentrations of $PM_{2.5}$ mass and major contributing species were often significantly different at the 95% confidence level. Greater values of $PM_{2.5}$ mass and individual species were generally reported from Speciation Trends Network methods than from the Desert Research Institute Sequential Filter Samplers. These differences can only be partially accounted for by known random errors. The authors concluded that the current uncertainty estimates used in the STN network may underestimate the actual uncertainty.

Positive Sampling Artifacts

The reaction of SO_2 (and other acid gases) with basic sites on glass fiber filters or with basic coarse particles on the filter leads to the formation of sulfate (or other nonvolatile salts, e.g., nitrate, chloride). These positive artifacts lead to the overestimation of total mass, and sulfate, and probably also nitrate concentrations. These problems were largely overcome by changing to quartz fiber or Teflon filters and by the separate collection of $PM_{2.5}$. However, the possible reaction of acidic gases with basic coarse particles remains a possibility, especially with PM_{10} and $PM_{10-2.5}$ measurements. These positive artifacts could be effectively eliminated by removing acidic gases in the sampling line with denuders coated with NaCl or Na_2CO_3.

Positive sampling artifacts also occur during measurement of particulate NH_4. The reaction of NH_3 with acidic particles (e.g. $2NH_3 + H_2SO_4 \rightarrow (NH_4)2SO_4$), either during sampling or during transportation, storage, and equilibration could lead to an overestimation of particulate

NH_4 concentrations. Techniques have been developed to overcome this problem: using a denuder to remove NH_3 during sampling and to protect the collected PM from NH_3 (Suh et al., 1992, 1994; Brauer et al., 1991; Koutrakis et al., 1988a,b; Keck and Wittmaack, 2006; Possanzini et al., 1999; Winberry et al., 1999). Hydrogen fluoride, citric acid, and phosphorous acids have been used as coating materials for the NH_3 denuder. Positive artifacts for particulate NH_4 can also be observed during sample handling due to contamination. No chemical analysis method, no matter how accurate or precise, can adequately represent atmospheric concentrations if the filters to which these methods are applied are improperly handled. Ammonia is emitted directly from human sweat, breath and smoking. It can then react with acidic aerosols on the filter to form ammonium sulfate, ammonium bisulfate and ammonium nitrate if the filter was not properly handled (Sutton el al., 2000). Therefore, it is important to keep filters away from ammonia sources, such as human breath, to minimize neutralization of the acidic compounds. Also, when filters are handled, preferably in a glove box, the analyst should wear gloves that are antistatic and powder-free to act as an effective contamination barrier.

Negative Sampling Artifact

Although sulfate is relatively stable on a Teflon filter, it is now well known that volatilization losses of particulate nitrates occur during sampling.

For nitrate, the effect on the accuracy of atmospheric particulate measurements from these volatilization losses is more significant for $PM_{2.5}$ than for PM_{10}. The FRM for $PM_{2.5}$ will likely suffer a loss of nitrates similar to that experienced with other simple filter collection systems. Sampling artifacts resulting from the loss of particulate nitrates represents a significant problem in areas such as southern California that experience high loadings of nitrates. Hering and Cass (1999) discussed errors in $PM_{2.5}$ mass measurements due to the volatilization of particulate nitrate. They examined data from two field measurement campaigns that were conducted in southern California: (1) the Southern California Air Quality Study (SCAQS) (Lawson, 1990) and (2) the 1986 CalTech study (Solomon et al., 1992). In both these studies, side-by-side sampling of $PM_{2.5}$ was conducted. One sampler collected particles directly onto a Teflon filter. The second sampler consisted of a denuder to remove gaseous HNO_3 followed by a nylon filter that absorbed the HNO_3 as it evaporated from $NITXNO_3$. In both studies, the denuder consisted of MgO-coated glass tubes (Appel et al., 1981). Fine particulate nitrate collected on the Teflon filter was compared to fine particulate nitrate collected on the denuded

nylon filter. In both studies, the $PM_{2.5}$ mass lost because of ammonium nitrate volatilization represented a significant fraction of the total $PM_{2.5}$ mass. The fraction of mass lost was higher during summer than during fall (17% versus 9% during the SCAQS study, and 21% versus 13% during the CalTech study). In regard to percentage loss of nitrate, as opposed to percentage loss of mass discussed above, Hering and Cass (1999) found that the amount of nitrate remaining on the Teflon filter samples was on average 28% lower than that on the denuded nylon filters.

Hering and Cass (1999) also analyzed these data by extending the evaporative model developed by Zhang and McMurry (1987). The extended model used by Hering and Cass (1999) takes into account the dissociation of collected particulate ammonium nitrate on Teflon filters into HNO_3 and NH_3 via three mechanisms: (1) the scrubbing of HNO_3 and NH_3 in the sampler inlet (John et al. (1988) showed that clean PM_{10} inlet surfaces serve as an effective denuder for HNO_3); (2) the heating of the filter substrate above ambient temperature by sampling; and (3) the pressure drop across the Teflon filter. For the sampling systems modeled, the flow-induced pressure drop was measured to be less than 0.02 atm, and the corresponding change in vapor pressure was 2%, so losses driven by pressure drop were not considered to be significant in this work. Losses from Teflon filters were found to be higher during the summer than during the winter, higher during the day compared to night, and reasonably consistent with modeled predictions.

Finally, during the SCAQS (Lawson, 1990) study, particulate samples also were collected using a Berner impactor and greased Tedlar substrates in size ranges from 0.05 to 10 μm in aerodynamic diameter. The Berner impactor $PM_{2.5}$ nitrate values were much closer to those from the denuded nylon filter than those from the Teflon filter, the impactor nitrate values being ~2% lower than the nylon filter nitrate for the fall measurements and ~7% lower for the summer measurements. When the impactor collection was compared to the Teflon filter collection for a nonvolatile species (sulfate), the results were in agreement. Chang et al. (2000) discuss reasons for reduced loss of nitrate from impactors.

Brook and Dann (1999) observed much higher nitrate losses during a study in which they measured particulate nitrate in Windsor and Hamilton, Ontario, Canada, by three techniques: (1) a single Teflon filter in a dichotomous sampler, (2) the Teflon filter in an annular denuder system (ADS), and (3) total nitrate including both the Teflon filter and the nylon back-up filter from the ADS. The Teflon filter from the dichotomous sampler averaged only 13% of the total

nitrate, whereas the Teflon filter from the ADS averaged 46% of the total nitrate. The authors concluded that considerable nitrate was lost from the dichotomous sampler filters during handling, which included weighing and X-ray fluorescence (XRF) measurement in a vacuum.

Kim et al. (1999) also examined nitrate-sampling artifacts by comparing denuded and non-denuded quartz and nylon filters during the PM_{10} Technical Enhancement Program (PTEP) in the South Coast Air Basin of California. They observed negative nitrate artifacts (losses) for most measurements; however, for a significant number of measurements, they observed positive nitrate artifacts. Kim et al. (1999) pointed out that random measurement errors make it difficult to measure true amounts of nitrate loss.

Diffusion denuder samplers, developed primarily to measure particle strong acidity (Koutrakis et al., 1988b, 1992), also can be used to study nitrate volatilization. Such techniques were used to measure loss of particulate nitrate from Teflon filters in seven U.S. cities (Babich et al., 2000). Measurements were made with two versions of the Harvard-EPA Annular Denuder System (HEADS). HNO_3 vapor was removed by a Na_2CO_3-coated denuder. Particulate nitrate was the sum of nonvolatile nitrate collected on a Teflon filter and volatized nitrate collected on a Na_2CO_3-coated filter downstream of the Teflon filter (full HEADS) or on a Nylon filter downstream of the Teflon filter (Nylon HEADS). It was found that the full HEADS (using a Na2CO3 filter) consistently underestimated the total particulate nitrate by approximately 20% compared to the nylon HEADS. Babich et al. (2000) found significant nitrate losses in Riverside, CA; Philadelphia, PA; and Boston, MA, but not in Bakersfield, CA; Chicago, IL; Dallas, TX; or Phoenix, AZ, where measurements were made only during the winter. Tsai and Huang (1995) used a diffusion denuder to study the positive and negative artifacts on glass and quartz filters. They found positive artifacts attributed to SO_2 and HNO_3 reaction with basic sites on glass fibers and basic particles and negative artifacts attributed to loss of HNO_3 and HCl due to volatilization of NH_4NO_3 and NH_4Cl and reaction of these species with acid sulfates.

Volatile compounds can also leave the filter after sampling and prior to filter weighing or chemical analysis. Losses of NO_3, NH_4, and Cl from glass and quartz-fiber filters that were stored in unsealed containers at ambient air temperatures for 2 to 4 weeks prior to analysis exceeded 50 percent (Witz et al., 1990). Storing filters in sealed containers and under refrigeration will minimize these losses.

Negative sampling artifacts due to decomposition and volatilization are also significant for particulate ammonium. Ammonium particulates, especially NH_4 N_3 nitrate NH_4 Cl are very sensitive to some environmental factors, such as temperature, relative humidity, acidity of aerosols, as well as to filter type (Spurny, 1999; Keck and Wittmaack, 2005). Any change in these parameters during the sampling period influences the position of the equilibrium between the particle phase and the gas phase. Keck and Wittmaack (2005) observed that at temperatures below 0C, acetate-nitrate, quartz fiber, and Teflon filters could properly collect particulate NH_4 NH_3 and Cl. At temperature above 0°C, the salts were lost from quartz fiber and Teflon filters, more so the higher the temperature and with no significant difference between quartz fiber and Teflon filters. The salts were lost completely from denuded quartz fiber filters above about 20C, and from non-undenuded quartz fiber and Teflon filters above about 25C. It is anticipated that current sampling techniques underestimate NH_4 concentrations due to the volatilization of NH_4, but fine particle mass contains many acidic compounds and consequently, a fraction of volatilized NH_4 (in the form of NH_3) can be retained on a PTFE filter by reaction with the acid compounds. Therefore, it is reasonable to assume that NH_4 loss will be less than the nitrate loss. Techniques have been applied to particulate ammonium sampling to correct particulate ammonium concentrations due to evaporation: a backup filter coated with hydrofluoric acids, citric acid, or phosphorous acids, is usually introduced to absorb the evaporated ammonium (as ammonia); the total ammonium concentration is the sum of the particle phase ammonium collected on the Teflon filter and the ammonia concentration collected on the backup filter.

Other Measurement Techniques

Nitrate

An integrated collection and vaporization cell was developed by Stolzenburg and Hering (2000) that provides automated, 10-min resolution monitoring of fine-particulate nitrate. In this system, particles are collected by a humidified impaction process and analyzed in place by flash vaporization and chemiluminescent detection of the evolved NO_x. In field tests in which the system was collocated with two FRM samplers, the automated nitrate sampler results followed the results from the FRM, but were offset lower. The system also was collocated with a HEADS and a SASS speciation sampler (MetOne Instruments). In all these tests, the automated sampler was well correlated to other samplers with slopes near 1 (ranging from 0.95 for the FRM to 1.06

for the HEADS) and correlation coefficients ranging from 0.94 to 0.996. During the Northern Front Range Air Quality Study in Colorado (Watson et al., 1998), the automated nitrate monitor captured the 12-min variability in fine-particle nitrate concentrations with a precision of approximately ± 0.5 $\mu g/m^3$ (Chow et al., 1998). A comparison with denuded filter measurements followed by ion chromatographic (IC) analysis (Chow and Watson, 1999) showed agreement within ± 0.6 $\mu g/m^3$ for most of the measurements, but exhibited a discrepancy of a factor of two for the elevated nitrate periods. More recent intercomparisons took place during the 1997 Southern California Ozone Study (SCOS97) in Riverside, CA. Comparisons with 14 days of 24-h denuder-filter sampling gave a correlation coefficient of $R^2 = 0.87$ and showed no significant bias (i.e., the regression slope is not significantly different from 1). As currently configured, the system has a detection limit of 0.7 $\mu g/m^3$ and a precision of 0.2 $\mu g/m^3$.

Sulfate

Continuous methods for the quantification of aerosol sulfur compounds first remove gaseous sulfur (e.g., SO_2, H_2S) from the sample stream by a diffusion tube denuder followed by the analysis of particulate sulfur (Cobourn et al., 1978; Durham et al., 1978; Huntzicker et al., 1978; Mueller and Collins, 1980; Tanner et al., 1980). Another approach is to measure total sulfur and gaseous sulfur separately by alternately removing particles from the sample stream. Particulate sulfur is obtained as the difference between the total and gaseous sulfur (Kittelson et al., 1978). The total sulfur content is measured by a flame photometric detector (FPD) by introducing the sampling stream into a fuel-rich, hydrogen-air flame (e.g., Stevens et al., 1969; Farwell and Rasmussen, 1976) that reduces sulfur compounds and measures the intensity of the chemiluminescence from electronically excited sulfur molecules (S2*). Because the formation of S2* requires two sulfur atoms, the intensity of the chemiluminescence is theoretically proportional to the square of the concentration of molecules that contain a single sulfur atom. In practice, the exponent is between 1 and 2 and depends on the sulfur compound being analyzed (Dagnall et al., 1967; Stevens et al., 1971). Calibrations are performed using both particles and gases as standards. The FPD can also be replaced by a chemiluminescent reaction with ozone that minimizes the potential for interference and provides a faster response time (Benner and Stedman, 1989, 1990). Capabilities added to the basic system include in situ thermal analysis and sulfuric acid speciation (Cobourn et al., 1978; Huntzicker et al., 1978; Tanner et al., 1980; Cobourn and Husar, 1982). Sensitivities for particulate sulfur as low as 0.1 $\mu g/m^3$, with time

resolution ranging from 1 to 30 min, have been reported. Continuous measurements of particulate sulfur content have also been obtained by on-line XRF analysis with resolution of 30 min or less (Jaklevic et al., 1981). During a field-intercomparison study of five different sulfur instruments, Camp et al. (1982) reported four out of five FPD systems agreed to within ± 5% during a 1-week sampling period.

AX2.9 POLICY RELEVANT BACKGROUND CONCENTRATIONS OF NITROGEN AND SULFUR OXIDES

Background concentrations of nitrogen and sulfur oxides used for purposes of informing decisions about NAAQS are referred to as Policy Relevant Background (PRB) concentrations. Policy Relevant Background concentrations are those concentrations that would occur in the United States in the absence of anthropogenic emissions in continental North America (defined here as the United States, Canada, and Mexico). Policy Relevant Background concentrations include contributions from natural sources everywhere in the world and from anthropogenic sources outside these three countries. Background levels so defined facilitate separation of pollution levels that can be controlled by U.S. regulations (or through international agreements with neighboring countries) from levels that are generally uncontrollable by the United States. EPA assesses risks to human health and environmental effects from NO_2 and SO_2 levels in excess of PRB concentrations.

Contributions to PRB concentrations include natural emissions of NO_2, SO_2, and photochemical reactions involving natural emissions of reduced nitrogen and sulfur compounds, as well as their long-range transport from outside North America. Natural sources of NO_2 and its precursors include biogenic emissions, wildfires, lightning, and the stratosphere. Natural sources of reduced nitrogen compounds, mainly NH_3, include biogenic emissions and wildfires. Natural sources of reduced sulfur species include anaerobic microbial activity in wetlands and volcanic activity. Volcanos and biomass burning are the major natural source of SO_2. Biogenic emissions from agricultural activities are not considered in the formation of PRB concentrations. Discussions of the sources and estimates of emissions are given in Section AX2.6.2.

Analysis of PRB Contribution to Nitrogen and Sulfur oxide Concentrations and Deposition over the United States

The MOZART-2 global model of tropospheric chemistry (Horowitz et al., 2003) is used to diagnose the PRB contribution to nitrogen and sulfur oxide concentrations, as well as to total (wet plus dry) deposition. The model setup for the present-day simulation has been published in a series of papers from a recent model intercomparison (Dentener et al., 2006a,b; Shindell et al., 2006; Stevenson et al., 2006; van Noije et al., 2006). MOZART-2 is driven by National Center for Environmental Prediction meteorological fields and IIASA 2000 emissions at a resolution of $1.9° \times 1.9°$ with 28 sigma levels in the vertical, and it includes gas- and aerosol phase chemistry. Results shown in Figures AX2-23 to AX2-27 are for the meteorological year 2001. Note that color images are available on the web. An additional "policy relevant background" simulation was conducted in which continental North American anthropogenic emissions were set to zero.

We first examine the role of PRB in contributing to NO_2 and SO_2 concentrations in surface air. Figure AX2-23 shows the annual mean NO_2 concentrations in surface air in the base case simulation (top panel) and the PRB simulation (middle panel), along with the percentage contribution of the background to the total base case NO_2 (bottom panel). Maximum concentrations in the base case simulation occur along the Ohio River Valley and in the Los Angeles basin. While present-day concentrations are often above 5 ppbv, PRB is less than 300 pptv over most of the continental United States, and less than 100 pptv in the eastern United States. The distribution of PRB (middle panel of Figure AX2-23) largely reflects the distribution of soil NO emissions, with some local enhancements due to biomass burning such as is seen in western Montana. In the northeastern United States, where present-day NO_2 concentrations are highest, PRB contributes <1% to the total.

The spatial pattern of present-day SO_2 concentrations over the United States is similar to that of NO_2, with highest concentrations (>5 ppbv) along the Ohio River valley (upper panel Figure AX2-24). Background SO_2 concentrations are orders of magnitude smaller, below 10 pptv over much of the United States (middle panel of Figure AX2-24). Maximum PRB concentrations of SO_2 are 30 ppt. In the Northwest where there are geothermal sources of SO_2, the contribution of PRB to total SO_2 is 70 to 80%. However, with the exception of the West Coast where volcanic SO_2 emissions enhance PRB concentrations, the PRB contributes <1% to present-day SO_2 concentrations in surface air (bottom panel Figure AX2-24).

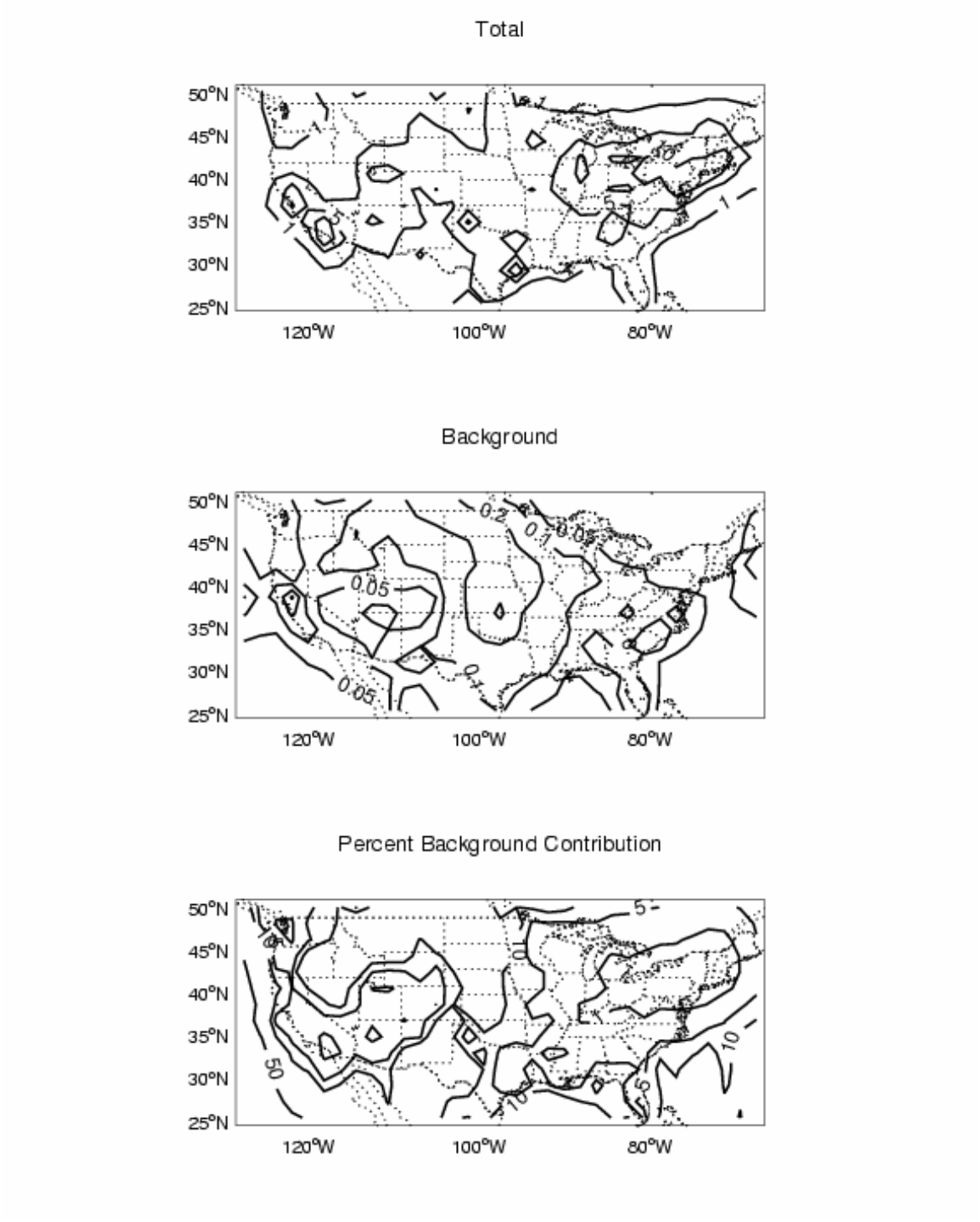


Figure AX2-23. **Annual mean concentrations of NO_2 (ppbv) in surface air over the United States in the present-day (upper panel) and policy relevant background (middle panel) MOZART-2 simulations. The bottom panel shows the percentage contribution of the background to the present-day concentrations. Please see text for details.**

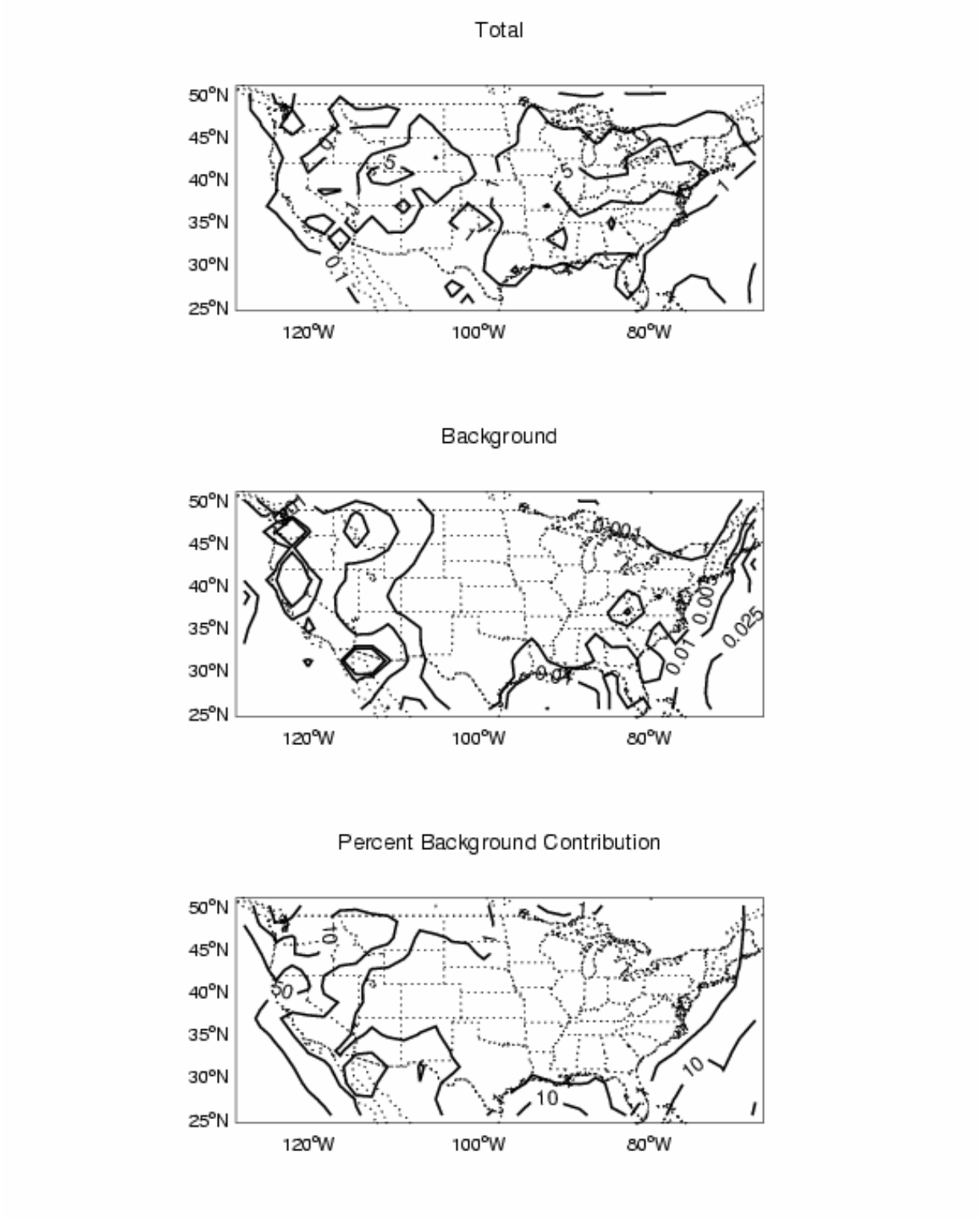


Figure AX2-24. Same as Figure AX2-23 but for SO_2 concentrations.

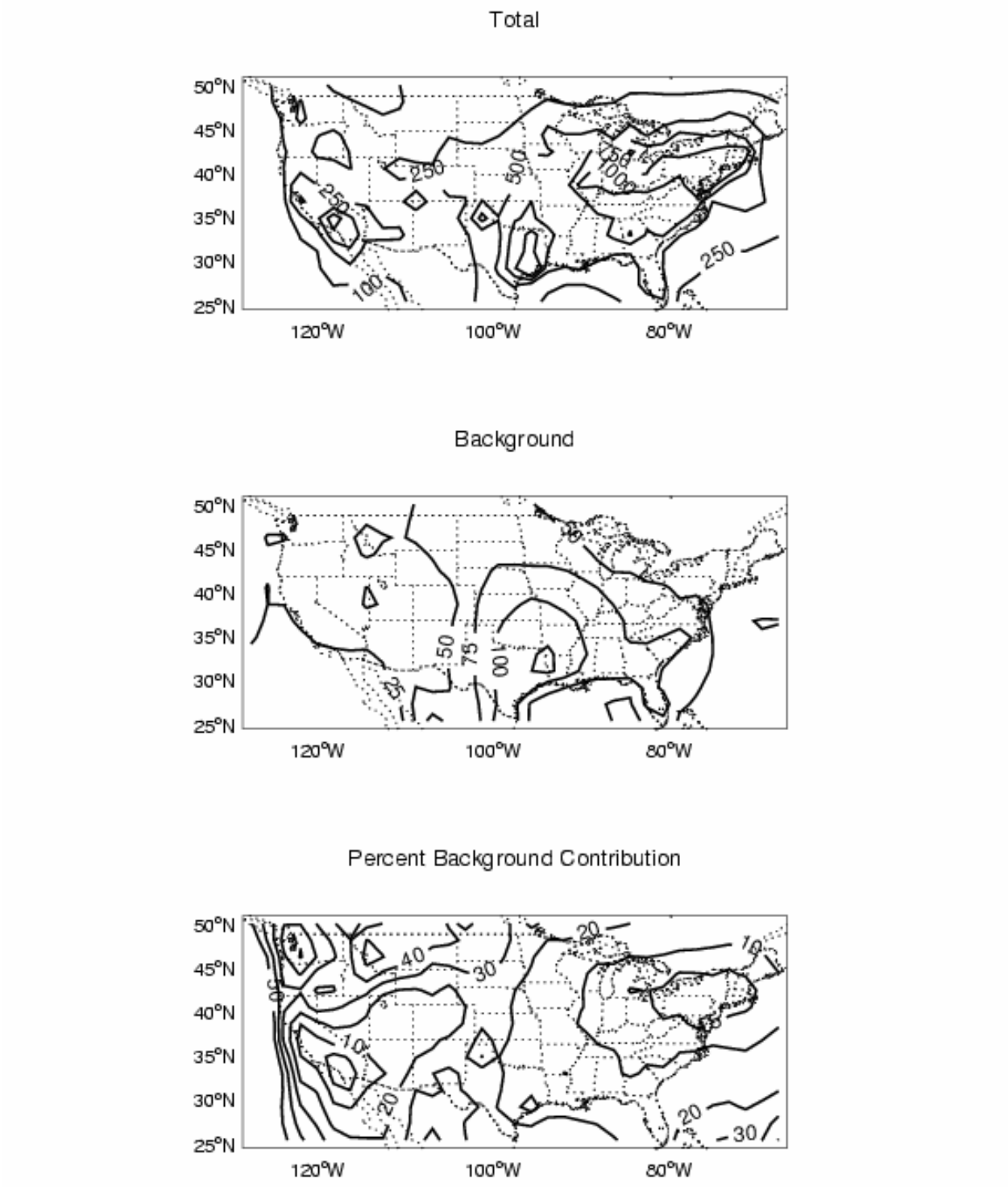


Figure AX2-25. Same as for Figure AX2-23 but for wet and dry deposition of HNO_3, NH_4NO_3, NO_x, HO_2NO_2, and organic nitrates (mg N $m^{-2}y^{-1}$).

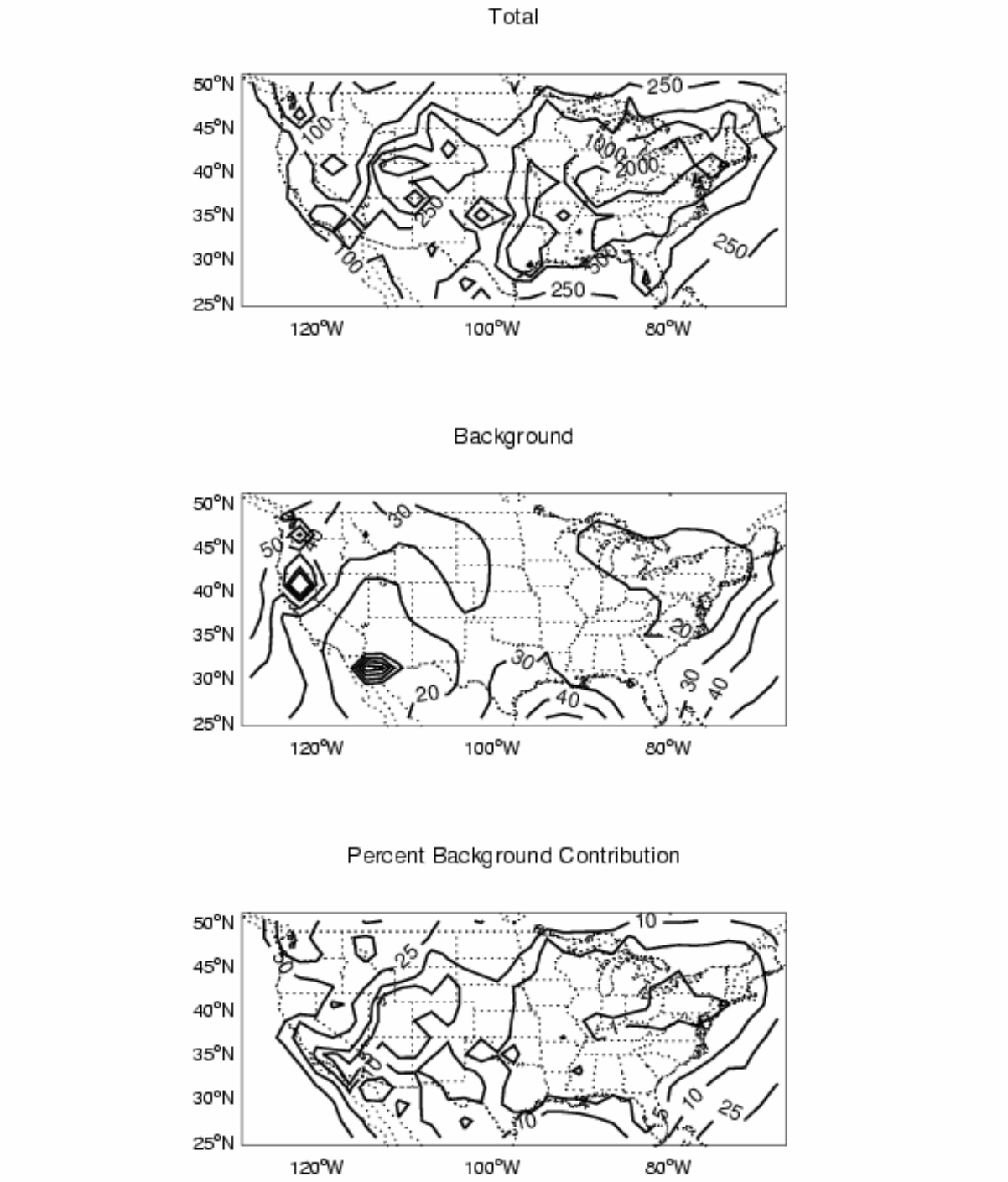


Figure AX2-26. Same as Figure AX2-23 but for SO_x deposition (SO_2 + SO_4) (mg S $m^{-2}y^{-1}$).

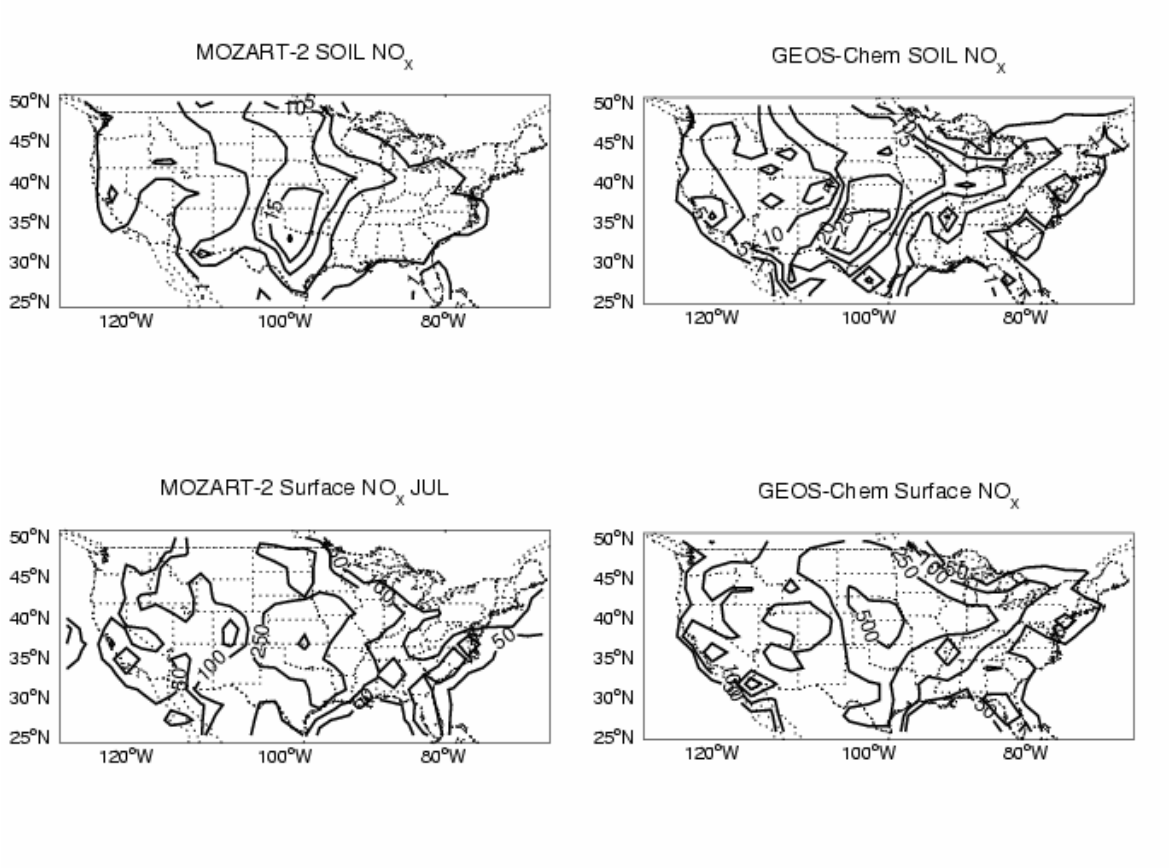


Figure AX2-27. July mean soil NO emissions (upper panels; 1×10^9 molecules cm^{-2} s^{1}) and surface PRB NO_x concentrations (lower panels; pptv) over the United States from MOZART-2 (left) and GEOS-Chem (right) model simulations in which anthropogenic O_3 precursor emissions were set to zero in North America.

The spatial pattern of NO_y (defined here as HNO_3, NH_4NO_3, NO_x, HO_2NO_2, and organic nitrates) wet and dry deposition is shown in Figure AX2-25. Figure AX2-25 (upper panel) shows that highest values are found in the eastern U.S. in and downwind of the Ohio River Valley. The pattern of nitrogen deposition in the PRB simulation (Figure AX2-25, middle panel), however, shows maximum deposition centered over Texas and in the Gulf Coast region, reflecting a combination of nitrogen emissions from lightning in the Gulf region, biomass burning in the Southeast, and from microbial activity in soils (maximum in central Texas and Oklahoma). The bottom panel of Figure AX2-25 shows that the PRB contribution to nitrogen

deposition is less than 20% over the eastern United States, and typically less than 50% in the western United States where NO_y deposition is low (25-50 mg N m^{-2} yr^{-1}).

Present-day SO_x (SO_2 + $SO_4^{=}$) deposition is largest in the Ohio River Valley, likely due to coal-burning power plants in that region, while background deposition is typically at least an order of magnitude smaller (Figure AX2-26). Over the eastern United States, the background contribution to SO_x deposition is <10%, and it is even smaller (<1%) where present-day SO_x deposition is highest. The contribution of PRB to sulfate deposition is highest in the western United States (>20%) because of geothermal sources of SO_2 and oxidation of dimethyl sulfide in the surface of the eastern Pacific.

Thus far, the discussion has focused on results from the MOZART-2 tropospheric chemistry model. In Figure AX2-27, results from MOZART-2 are compared with those from another tropospheric chemistry model, GEOS-Chem (Bey et al., 2001), which was previously used to diagnose PRB O_3 (Fiore et al., 2003; U.S. EPA, 2006). In both models, the surface PRB NO_x concentrations tend to mirror the distribution of soil NO emissions, which are highest in the Midwest. The higher soil NO emissions in GEOS-Chem (by nearly a factor of 2) as compared to MOZART-2 reflect different assumptions regarding the contribution to soil NO emissions largely through fertilizer, since GEOS-Chem total soil NO emissions are actually higher than MOZART-2 (0.07 versus 0.11 Tg N) over the United States in July. Even with the larger PRB soil NO emissions, surface NO_x concentrations in GEOS-Chem are typically below 500 pptv.

It is instructive to also consider measurements of SO_2 at relatively remote monitoring sites, i.e., site located in sparsely populate areas not subject to obvious local sources of pollution. Berresheim et al. (1993) used a type of atmospheric pressure ionization mass spectrometer (APIMS) at Cheeka Peak, WA (48.30N 124.62W, 480 m asl), in April 1991 during a field study for DMS oxidation products. Sulfur Dioxide concentrations ranged between 20 and 40 pptv. Thornton et al. (2002) have also used an APIMS with an isotopically labeled internal standard to determine background SO_2 levels. SO_2 concentrations of 25 to 40 pptv were observed in northwestern Nebraska in October 1999 at 150m above ground using the NCAR C-130 (Thornton, unpublished data). These data are comparable to remote central south Pacific convective boundary layer SO_2 (Thornton et al., 1999).

Volcanic sources of SO_2 in the US are limited to the Pacific Northwest, Alaska, and Hawaii. Since 1980 the Mt. St. Helens volcano in Washington Cascade Range (46.20 N,

122.18 W, summit 2549 m asl) has been a variable source of SO_2. Its major impact came in the explosive eruptions of 1980, which primarily affected the northern part of the mountain west of the US. The Augustine volcano near the mouth of the Cook Inlet in southwestern Alaska (59.363 N, 153.43 W, summit 1252 m asl) has had SO_2 emissions of varying extents since its last major eruptions in 1986. Volcanoes in the Kamchatka peninsula of eastern region of Siberian Russia do not particularly impact the surface concentrations in the northwestern NA. The most serious impact in the US from volcanic SO_2 occurs on the island of Hawaii. Nearly continuous venting of SO_2 from Mauna Loa and Kilauea produce SO_2 in such large amounts so that >100 km downwind of the island SO_2 concentrations can exceed 30 ppbv (Thornton and Bandy, 1993). Depending on the wind direction the west coast of Hawaii (Kona region) has had significant impacts from SO_2 and acidic sulfate aerosols for the past decade. Indeed, SO_2 levels in Volcanoes National Park, HI exceeded the 3-h and the 24-h NAAQS in 2004 -2005. The area's design value is 0.6 ppm for the 3-h, and 0.19 ppm for the 24-h NAAQS (U.S. EPA, 2006).

Overall, the background contribution to nitrogen and sulfur oxides over the United States is relatively small, except for SO_2 in areas where there is volcanic activity.

TABLE AX2-1. ATMOSPHERIC LIFETIMES OF SULFUR DIOXIDE AND REDUCED SULFUR SPECIES WITH RESPECT TO REACTION WITH OH, NO_3, AND CL RADICALS

	OH		NO_3		Cl	
Compound	**$k \times 10^{12}$**	**τ**	**$k \times 10^{12}$**	**τ**	**$k \times 10^{12}$**	**τ**
SO_2	1.6	7.2d	NA		NA	
CH_3-S-CH_3	5.0	2.3 d	1.0	1.1 h	400	29 d
H_2S	4.7	2.2 d	NA		74	157 d
CS_2	1.2	9.6 d	<0.0004	> 116 d	<0.004	NR
OCS	0.0019	17 y	<0.0001	> 1.3 y	<0.0001	NR
CH_3-S-H	33	8.4 h	0.89	1.2 h	200	58 d
CH_3-S-S-CH_3	230	1.2 h	0.53	2.1 h	NA	

Notes:

NA = Reaction rate coefficient not available. NR = Rate coefficient too low to be relevant as an atmospheric loss mechanism. Rate coefficients were calculated at 298 K and 1 atmosphere.

y = year. d = day. h = hour. OH = 1×10^6/cm^3; NO_3 = 2.5×10^8/cm^3; Cl = 1×10^3/cm^3.

[1] Rate coefficients were taken from JPL Chemical Kinetics Evaluation No. 14 (JPL, 2003).

TABLE AX2-2A. RELATIVE CONTRIBUTIONS OF VARIOUS REACTIONS TO THE TOTAL S(IV) OXIDATION RATE WITHIN A SUNLIT CLOUD, 10 MINUTES AFTER CLOUD FORMATION

Reaction	% of Total [a]	% of Total [b]
Gas Phase		
$OH + SO_2$	3.5	3.1
Aqueous Phase		
$O_3 + HSO_3^-$	0.6	0.7
$O_3 + SO_3^{2-}$	7.0	8.2
$H_2O_2 + SO_3^-$	78.4	82.1
$CH_3OOH + HSO_3^-$	0.1	0.1
$HNO_4 + HSO_3^-$	9.0	4.4
$HOONO + HSO_3^-$	<0.1	<0.1
$HSO_5^- + HSO_3^-$	1.2	<0.1
$SO_5^- + SO_3^{2-}$	<0.1	<0.1
$HSO_5^- + Fe^{2+}$		0.6

[a] In the absence of transition metals.

[b] In the presence of iron and copper ions.

Source: Adapted from Warneck (1999).

TABLE AX2-2B. RELATIVE CONTRIBUTIONS OF VARIOUS GAS AND AQUEOUS PHASE REACTIONS TO AQUEOUS NITRATE FORMATION WITHIN A SUNLIT CLOUD, 10 MINUTES AFTER CLOUD FORMATION

Reaction	% of Total [a]	% of Total [b]
Gas Phase		
$OH + NO_2 + M$	57.7	67.4
Aqueous Phase		
$N_2O_{5g} + H_2O$	8.1	11.2
$NO_3 + Cl^-$	<0.1	0.1
$NO_3 + HSO_3^-$	0.7	1.0
$NO_3 + HCOO^-$	0.6	0.8
$HNO_4 + HSO_3^-$	31.9	20.5
$HOONO + NO_3^-$	0.8	<0.1
$O_3 + NO_2^-$	<0.1	<0.1

[a] In the absence of transition metals.

[b] In the presence of iron and copper ions.

Source: Adapted from Warneck (1999).

TABLE AX2-3. EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
Source Category			
TOTAL ALL SOURCES	23.19	4.08	16.87
FUEL COMBUSTION TOTAL	9.11	0.02	14.47
FUEL COMB. ELEC. UTIL.	5.16	<0.01	11.31
Coal	4.50	<0.01	10.70
Bituminous	2.90		8.04
Subbituminous	1.42		2.14
anthracite & lignite	0.18		0.51
Other	<0.01		
Oil	0.14	<0.01	0.38
Residual	0.13		0.36
Distillate	0.01		0.01
Gas	0.30	<0.01	0.01
Natural	0.29		
Process	0.01		
Other	0.05	<0.01	0.21
Internal Combustion	0.17	<0.01	0.01
FUEL COMBUSTION INDUSTRIAL	3.15	<0.01	2.53
Coal	0.49	<0.01	1.26
Bituminous	0.25		0.70
Subbituminous	0.07		0.10
anthracite & lignite	0.04		0.13
Other	0.13		0.33
Oil	0.19	<0.01	0.59
Residual	0.09		0.40
Distillate	0.09		0.16
Other	0.01		0.02
Gas	1.16	<0.01	0.52
Natural	0.92		
Process	0.24		
Other	<0.01		
Other	0.16	<0.01	0.15
wood/bark waste	0.11		
liquid waste	0.01		
Other	0.04		
Internal Combustion	1.15	<0.01	0.01

TABLE AX2-3 (cont'd). EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
FUEL COMB. OTHER	0.80	<0.01	0.63
Commercial/Institutional Coal	0.04	<0.01	0.16
Commercial/Institutional Oil	0.08	<0.01	0.28
Commercial/Institutional Gas	0.25	<0.01	0.02
Misc. Fuel Comb. (Except Residential)	0.03	<0.01	0.01
Residential Wood	0.03		<0.01
Residential Other	0.36		0.16
distillate oil	0.06		0.15
bituminous/subbituminous coal	0.26		<0.01
Other	0.04		<0.01
INDUSTRIAL PROCESSES TOTAL	1.10	0.21	1.54
CHEMICAL & ALLIED PRODUCT MFG	0.12	0.02	0.36
Organic Chemical Mfg	0.02	<0.01	0.01
Inorganic Chemical Mfg	0.01	<0.01	0.18
sulfur compounds			0.17
Other			0.02
Polymer & Resin Mfg	<0.01	<0.01	<0.01
Agricultural Chemical Mfg	0.05	0.02	0.05
ammonium nitrate/urea mfg.		<0.01	
Other		0.02	
Paint, Varnish, Lacquer, Enamel Mfg	0.00		0.00
Pharmaceutical Mfg	0.00		0.00
Other Chemical Mfg	0.03	<0.01	0.12
METALS PROCESSING	0.09	<0.01	0.30
Non-Ferrous Metals Processing	0.01	<0.01	0.17
Copper			0.04
Lead			0.07
Zinc			0.01
Other			<0.01
Ferrous Metals Processing	0.07	<0.01	0.11
Metals Processing NEC	0.01	<0.01	0.02

TABLE AX2-3 (cont'd). EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
PETROLEUM & RELATED INDUSTRIES	0.16	<0.01	0.38
Oil & Gas Production	0.07	<0.01	0.11
natural gas			0.11
Other			0.01
Petroleum Refineries & Related Industries	0.05	<0.01	0.26
fluid catalytic cracking units		<0.01	0.16
Other		<0.01	0.07
Asphalt Manufacturing	0.04		0.01
OTHER INDUSTRIAL PROCESSES	0.54	0.05	0.46
Agriculture, Food, & Kindred Products	0.01	<0.01	0.01
Textiles, Leather, & Apparel Products	<0.01	<0.01	<0.01
Wood, Pulp & Paper, & Publishing Products	0.09	<0.01	0.10
Rubber & Miscellaneous Plastic Products	<0.01	<0.01	<0.01
Mineral Products	0.42	<0.01	0.33
cement mfg	0.24		0.19
glass mfg	0.01		
Other	0.10		0.09
Machinery Products	<0.01	<0.01	<0.01
Electronic Equipment	<0.01	<0.01	<0.01
Transportation Equipment	<0.01		<0.01
Miscellaneous Industrial Processes	0.01	0.05	0.02
SOLVENT UTILIZATION	0.01	<0.01	<0.01
Degreasing	<0.01	<0.01	<0.01
Graphic Arts	<0.01	<0.01	<0.01
Dry Cleaning	<0.01	<0.01	<0.01
Surface Coating	<0.01	<0.01	<0.01
Other Industrial	<0.01	<0.01	<0.01
Nonindustrial	<0.01		
Solvent Utilization NEC	<0.01		

TABLE AX2-3 (cont'd). EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
STORAGE & TRANSPORT	<0.01	<0.01	0.01
Bulk Terminals & Plants	<0.01	<0.01	<0.01
Petroleum & Petroleum Product Storage	<0.01	<0.01	<0.01
Petroleum & Petroleum Product Transport	<0.01	<0.01	<0.01
Service Stations: Stage II	<0.01		<0.01
Organic Chemical Storage	<0.01	<0.01	<0.01
Organic Chemical Transport	0.01		<0.01
Inorganic Chemical Storage	<0.01	<0.01	<0.01
Inorganic Chemical Transport	<0.01		<0.01
Bulk Materials Storage	0.01	<0.01	<0.01
WASTE DISPOSAL & RECYCLING	0.17	0.14	0.03
Incineration	0.06	<0.01	0.02
Industrial			
Other			<0.01
Open Burning	0.10	<0.01	<0.01
Industrial			<0.01
land clearing debris			
Other			<0.01
POTW	<0.01	0.14	<0.01
Industrial Waste Water	<0.01	<0.01	<0.01
TSDF	<0.01	<0.01	<0.01
Landfills	<0.01	<0.01	<0.01
Industrial			<0.01
Other			<0.01
Other	<0.01	<0.01	<0.01

TABLE AX2-3 (cont'd). EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
TRANSPORTATION TOTAL	12.58	0.32	0.76
HIGHWAY VEHICLES	8.09	0.32	0.30
Light-Duty Gas Vehicles & Motorcycles	2.38	0.20	0.10
light-duty gas vehicles	2.36		0.10
Motorcycles	0.02		0.00
Light-Duty Gas Trucks	1.54	0.10	0.07
light-duty gas trucks 1	1.07		0.05
light-duty gas trucks 2	0.47		0.02
Heavy-Duty Gas Vehicles	0.44	<0.01	0.01
Diesels	3.73	<0.01	0.12
heavy-duty diesel vehicles	3.71		
light-duty diesel trucks	0.01		
light-duty diesel vehicles	0.01		
OFF-HIGHWAY	4.49	<0.01	0.46
Non-Road Gasoline	0.23	<0.01	0.01
Recreational	0.01		
Construction	0.01		
Industrial	0.01		
lawn & garden	0.10		
Farm	0.01		
light commercial	0.04		
Logging	<0.01		
airport service	<0.01		
railway maintenance	<0.01		
recreational marine vessels	0.05		
Non-Road Diesel	1.76	<0.01	0.22
Recreational	0.00		
Construction	0.84		
Industrial	0.15		
lawn & garden	0.05		
Farm	0.57		
light commercial	0.08		
Logging	0.02		
airport service	0.01		
railway maintenance	<0.01		
recreational marine vessels	0.03		

TABLE AX2-3 (cont'd). EMISSIONS OF NITROGEN OXIDES, AMMONIA, AND SULFUR DIOXIDE IN THE UNITED STATES IN 2002

2002 Emissions (Tg/year)	NO_x [1]	NH_3	SO_2
Aircraft	0.09		0.01
Marine Vessels	1.11		0.18
Diesel	1.11		
residual oil			
Other			
Railroads	0.98		0.05
Other	0.32	<0.01	0.00
liquefied petroleum gas	0.29		
compressed natural gas	0.04		
MISCELLANEOUS	0.39	3.53	0.10
Agriculture & Forestry	<0.01	3.45	<0.01
agricultural crops		<0.01	
agricultural livestock		2.66	
Other Combustion		0.08	0.10
Health Services			
Cooling Towers			
Fugitive Dust			
Other			
Natural Sources	3.10	0.03	

[1] Emissions are expressed in terms of NO_2.

[2] Estimate based on Guenther et al. (2000).

Source: U.S. Environmental Protection Agency (2006).

TABLE AX2-3. SATELLITE INSTRUMENTS USED TO RETRIEVE TROPOSPHERIC NO_2 COLUMNS.

Instrument	Coverage	Typical U.S. Measurement Time	Typical Resolution (km)	Return Time (days)[1]	Instrument Overview
GOME	1995-2002	10:30-11:30 AM	320 × 40	3	Burrows et al. (1999)
SCIAMACHY	2002-	10:00-11:00 AM	30 × 60	6	Bovensmann et al. (1999)
OMI	2004-	12:45-1:45 PM	13 × 24	1	Levelt et al. (2006)

[1] Return time is reported here for cloud free conditions. Note that due to precession of the satellite's orbit, return measurements are close to but not made over the same location. In practice, clouds decrease observation frequency by a factor of 2.

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AX3. CHAPTER 3 ANNEX – AMBIENT CONCENTRATIONS AND EXPOSURES

AX3.1 INTRODUCTION

Topics discussed in this chapter include the characterization of ambient air quality for nitrogen dioxide (NO_2), the uses of these data in assessing human exposures to NO_2; concentrations and sources of NO_2 in different microenvironments, and personal exposures to NO_2. The NO_2 data contained in this chapter are taken mainly from the U.S. Environmental Protection Agency's Air Quality System (AQS) database (formerly the AIRS database) (U.S. Environmental Protection Agency, 2007).

Characterizing Ambient NO_2 Concentrations

The "concentration" of a specific air pollutant is typically defined as the amount (mass) of that material per unit volume of air. However, most of the data presented in this chapter are expressed as "mixing ratios" in terms of a volume-to-volume ratio (e.g., parts per million [ppm] or parts per billion [ppb]. Data expressed this way are often referred to as concentrations, both in the literature and in the text, following common usage. Human exposures are expressed in units of mixing ratio times time.

Relationship to the 1993 Air Quality Criteria Document for Nitrogen Oxides

The 1993 AQCD for Oxides of Nitrogen emphasized NO_2 indoor sources (gas stoves) and the relationship between personal total exposure and indoor or outdoor NO_2 concentrations. At that time, only few personal exposure studies had been conducted with an emphasis on residential indoor NO_2 sources and concentrations. Although the concept of microenvironment had been introduced in the document, NO_2 concentrations were seldom reported for microenvironments other than residences. Exposure measurements at that time relied on Palmes tubes and Yanagisawa badges; and exposure-modeling techniques were limited mainly to simple linear regression. In the 1993 AQCD, NO_2 was treated as an independent risk factor, and confounding issues were not mentioned in the human environmental exposure chapters.

The current chapter summarizes and discusses the state-of-the-science and technology regarding NO_2 human exposures since 1993. Since then, numerous human exposure studies

have been conducted with new measurement and modeling techniques. Microenvironmental measurements were not limited to residential indoor environments; NO_2 concentrations were also measured in vehicles, schools and offices, and microenvironments close to traffic. More indoor sources have been identified and more NO_2 formation and transformation mechanisms in the indoor environment have been reported. Both indoor and outdoor NO_2 have been treated as components of a pollutant mixture, and therefore the concepts of confounding and surrogacy have been discussed in the current chapter.

AX3.2 AMBIENT CONCENTRATIONS OF NITROGEN OXIDES AND RELATED SPECIES

As discussed in Chapter 2, most measurements of NO_x are made by instruments that convert NO_2 to NO, which is then measured by chemiluminescence. However, the surface converters that reduce NO_2 to NO also reduce other reactive NO_y species. As indicated in Chapter 2, NO_y compounds consist of NO_x, gas phase inorganic nitrates, such as $ClNO_3$; organic nitrates, such as PANs; inorganic acids, given by the formulas HNO_y (y = 2 to 4); and particulate nitrate. In urban areas or in rural areas where there are large local sources, NO and NO_2 are expected to be the major forms of NO_y. Thus, interference from PANs and other NO_y species near sources are expected to minor; in most rural and remote areas, interference may be substantial as concentrations of other NO_y species may be much larger than those for NO and NO_2 (National Research Council, 1991). Examples will be presented in Section AX3.3.5.

Data for NO_x in addition to NO_2 is reported into the U.S. Environmental Protection Agency's Air Quality System (AQS), but data for NO is not reported, even though measurements of NO are not affected by artifacts caused by products of NO_2 oxidation and therefore should be the most reliable. By definition, NO_x is equal to the sum of NO and NO_2, so the concentration of NO can be found by subtraction. However, measurements are obtained for NO and NO_x every 2 to 3 min, but hourly averages for NO_2 and NO_x are reported into AQS. The locations of NO_2 monitoring sites are shown in Figure AX3.1. As can be seen from Figure AX3.1, there are large areas of the United States for which data for ambient NO_2 are not collected. The percentile distribution of NO_2 concentrations in urban and nonurban areas in the U.S. for different averaging periods is shown in Table AX3.1.

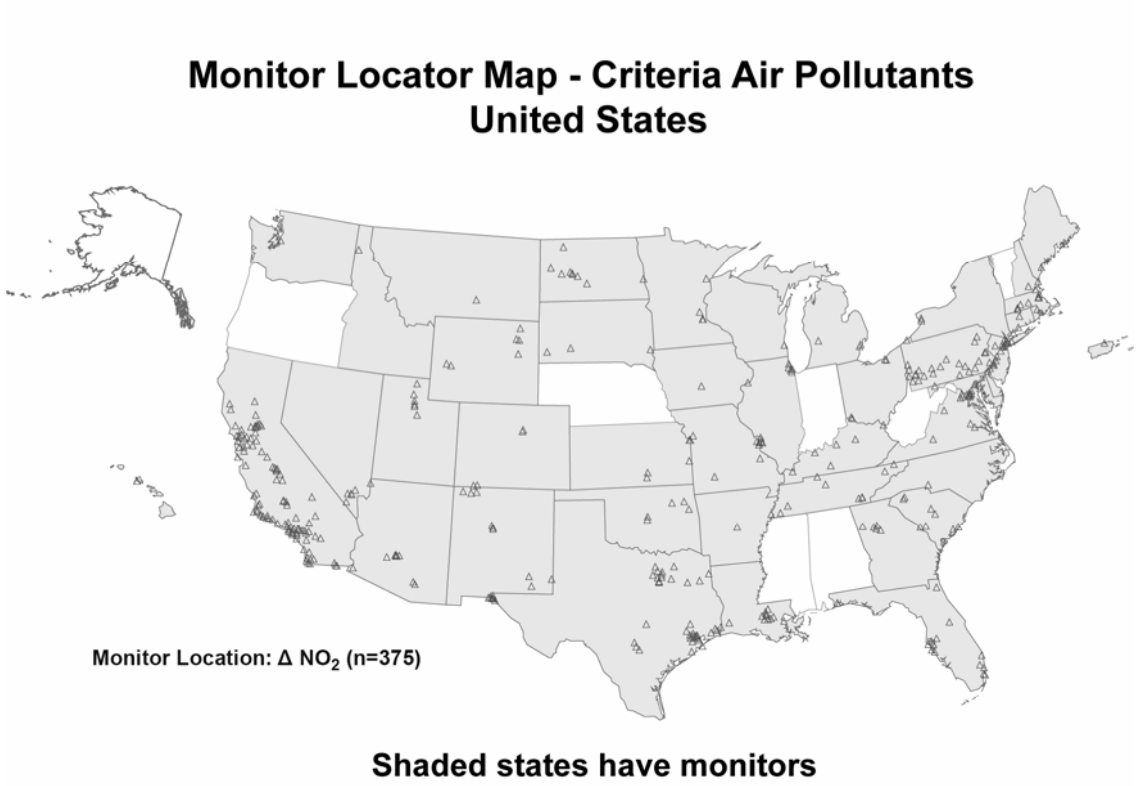

Figure AX3.1. Location of ambient NO_2 monitors in the United States.

Because of their short lifetime with respect to oxidation to PANs and HNO_3, NO_x concentrations are highly spatially and temporally variable. Average concentrations range from tens of ppt in remote areas of the globe to tens of ppb in urban cores, i.e., by three orders of magnitude. Median NO, NO_x, and NO_y concentrations at the surface are typically below 0.01, 0.05, and 0.3 ppb, respectively, in remote areas such as Alaska, northern Canada, and the eastern Pacific; median NO_y concentrations range from about 0.7 to about 4.3 ppb at regional background sites in the eastern United States (Emmons et al., 1997). Note that the last two values, especially, contain a substantial contribution from pollution. Maximum short-term average (1-h) NO_x concentrations near heavy traffic (e.g., in Los Angeles, CA) approach 1 ppm, but these levels decrease rapidly away from sources. Even at sites where such high hourly values are found, 24-h average concentrations are much lower. For example, the maximum 24-h average NO_x concentration at any site in Los Angeles in 2004 was 82 ppb.

NO_2 concentrations are likewise highly spatially and temporally variable. The overall annual mean concentration of NO_2 at U.S. monitoring sites is about 15 ppb. Most sites monitoring NO_2 are located in populated areas and values outside of urban and suburban areas can be much lower. Perhaps the most comprehensive characterization of ambient NO_2 levels has been performed by the California Air Resources Board (CARB) as part of the review of the air quality standards for California (CARB, 2007). On a statewide basis, the average NO_2 concentration was about 15 ppb from 2002 to 2004. Highest average values of about 27 ppb were found in the South Coast Air Basin. The maximum 1-h average NO_2 concentration during the same period was 262 ppb, again in the South Coast Air Basin. However, maximum 1-h concentrations of NO_2 were about 150 ppb in Los Angeles, CA in 2004, implying that the high NO_x level (~1ppm) cited above for Los Angeles consisted mainly of NO. It is highly unlikely that NO_x oxidation products constituted a significant fraction of the NO_x reported.

AX3.2.1 Spatial and Temporal Variability in Ambient Concentrations of NO_2 and Related Species in Urban Areas

As noted earlier, the number of monitoring sites reporting data for NO_2 is considerably smaller than for other criteria pollutants. As a result, there are few urban areas where there exist sufficient data to evaluate the spatial variability in NO_2 even though most of the NO_2 monitors are found in urban or suburban areas. Analyses of spatial variability in NO and NO_2 are thus limited to Los Angeles, CA and Chicago, IL. Also, as noted in Chapter 2, current methods for measuring NO_2 are subject to interference from its oxidation products. Hence the reported values represent upper limits for the true NO_2 concentration. Near highways or other NO_x sources, the measurements should give more accurate values, but because of variability in the time needed for conversion of NO_x to NO_z, no firm rules can be applied to account for the presence of NO_z species such as HNO_3 and PANs. These considerations introduce additional uncertainty into the interpretation of any metrics (e.g., correlation coefficients, concentration differences) that are used to characterize spatial variability in NO_2 concentrations.

The spatial variability in 1 h average NO_2 concentrations in New York, NY; Atlanta, GA; Chicago, IL; Houston, TX; Los Angeles, CA; and Riverside, CA is characterized in this section. These areas were chosen to provide analyses to help guide risk assessment and to provide a general overview of the spatial variability of NO_2 in cities where health outcome studies have been conducted. Statistical analyses of the human health effects of airborne pollutants based on

aggregate population time-series data have often relied on ambient concentrations of pollutants measured at one or more central sites in a given metropolitan area. In particular, cities with low traffic densities that are located downwind of major sources of precursors are heavily influenced by long range transport and tend to show smaller spatial variability (e.g., New Haven, CT) than those source areas with high traffic densities located upwind (e.g., New York, NY).

Metrics for characterizing spatial variability include the use of Pearson correlation coefficients, values of the 90th percentile (P90) of the absolute difference in concentrations, and coefficients of divergence (COD) The COD is defined as follows:

$$COD_{jk} = \sqrt{\frac{1}{p}\sum_{i=1}^{p}\left(\frac{X_{ij} - X_{ik}}{X_{ij} + X_{ik}}\right)^2} \qquad \text{(AX3-1)}$$

where x_{ij} and x_{ik} represent observed concentrations averaged over some measurement averaging period (hourly, daily, etc.), for measurement period *i* at site *j* and site *k* and *p* is the number of observations. These methods of analysis follow those used for characterizing $PM_{2.5}$ and $PM_{10-2.5}$ concentrations in Pinto et al. (2004) and in the latest edition of the PM AQCD (U.S. Environmental Protection Agency, 2004a).

Summary statistics for the spatial variability in several urban areas across the United States are shown in Table AX3.2. These areas were chosen because they are the major urban areas with at least five monitors operating from 2003 to 2005. Values in parentheses below the city name refer to the number of sites colleting data. The second column shows the mean 1 h average concentration across all sites and the range in means at individual sites. The third column gives the range of Pearson correlation coefficients between individual site pairs in the urban area. The fourth column shows the 90th percentile absolute difference in concentrations between site pairs. The fifth column gives the coefficient of divergence (COD).

As can be seen from the table, mean concentrations at individual sites vary by factors of 1.5 to 6 in the MSAs examined. Correlations between individual site pairs range from slightly negative to highly positive in a given urban area. The sites in New York City tend to be the most highly correlated and show the highest mean levels, reflecting their proximity to traffic, as evidenced by the highest mean concentration of all the entries. However, correlation coefficients are not sufficient for describing spatial variability as concentrations at two sites may be highly correlated but show differences in levels. Thus, the range in mean concentrations is given. Even

in New York City, the spread in mean concentrations is about 40% of the city-wide mean (12/29). The relative spread in mean concentrations is larger in the other urban areas shown in Table AX3.2. As might be expected, the 90th percentile concentration spreads are even larger than the spreads in the means.

The same statistics shown in Table AX3.2 have been used to describe the spatial variability of $PM_{2.5}$ (U.S. Environmental Protection Agency, 2004; Pinto et al., 2005) and O_3 (U.S. Environmental Protection Agency, 2006). However, because of relative sparseness in data coverage for NO_2, spatial variability in all cities that were considered for $PM_{2.5}$ and O_3 could not be considered here. Thus, the number of cities included below is much smaller than for either O_3 (24 urban areas) or $PM_{2.5}$ (27 urban areas). Even in those cities where there are monitors for all three pollutants, data may not have been collected at the same locations and even if they were, there would be variable influence from local sources. For example, concentrations of NO_2 collected near traffic will be highest in an urban area, but concentrations of O_3 will tend to be lowest because of titration by NO forming NO_2. However, some general observations can still be made. Mean concentrations of NO_2 at individual monitoring sites are not as highly variable as for O_3 but are more highly variable than $PM_{2.5}$. Lower bounds on inter-site correlation coefficients for $PM_{2.5}$ and for O_3 tend to be much higher than NO_2 in the same areas shown in Table AX3.2. CODs for $PM_{2.5}$ are much lower than for O_3, whereas CODs for NO_2 tend to be the largest among the three pollutants. Therefore, it is apparent that there is the potential for errors from the use of ambient monitors to characterize exposures either at the community or personal level, and that this potential may be higher than for either O_3 or $PM_{2.5}$.

Small Scale Vertical Variability

Inlets to instruments for monitoring gas phase criteria pollutants can be located from 3 to 15 m above ground level (CFR 58, Appendix E, 2002). Depending on the pollutant, either there can be positive, negative or no vertical gradient from the ground to the monitor inlet. Pollutants that are formed over large areas by atmospheric photochemical reactions and are destroyed by deposition to the surface or by reaction with pollutants emitted near the surface show positive vertical gradients. Pollutants that are emitted by sources at or just above ground level show negative vertical gradients. Pollutants with area sources and have minimal deposition velocities

show little or no vertical gradient. Restrepo et al. (2004) compared data for criteria pollutants collected at fixed monitoring sites at 15 m above the surface on a school rooftop to those measured by a van whose inlet was 4 m above the surface at monitoring sites in the South Bronx during two sampling periods in November and December 2001. They found that CO, SO_2, and NO_2 showed positive vertical gradients, whereas O_3 showed a negative vertical gradient and $PM_{2.5}$ showed no significant vertical gradient. As shown in Figure AX3.2, NO_2 mixing ratios obtained at 4 m (mean ~74 ppb) were about a factor of 2.5 higher than at 15 m (mean ~30 ppb). Because tail pipe emissions occur at lower heights, NO_2 values could have been much higher nearer to the surface, and the underestimation of NO_2 values by monitoring at 15 m even larger. Restrepo et al. (2004) note that the use of the NO_2 data obtained by the stationary monitors would result in an underestimate of human exposures to NO_2 in the South Bronx. However, this issue is most likely not unique to the South Bronx and could arise in other large urban areas in the U.S. with populations of similar demographic and socioeconomic characteristics.

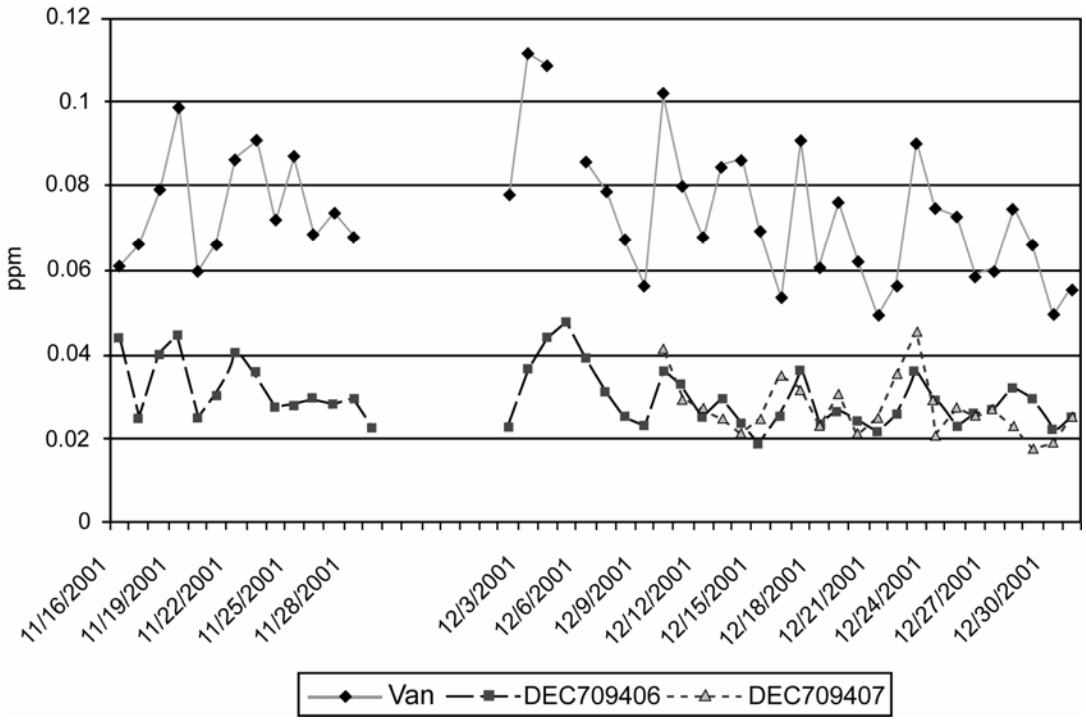

Figure AX3.2. NO_2 concentrations measured at 4 m (Van) and at 15 m at NY Department of Environmental Conservation sites (DEC709406 and DEC709407).

Source: Restrepo et al. (2004).

AX3.2.2 Temporal Variability in Nitrogen Oxides

AX3.2.2.1 Diurnal Variability in NO_2 Concentrations

As might be expected from a pollutant having a major traffic source, the diurnal cycle of NO_2 in typical urban areas is characterized by traffic emissions, with peaks in emissions occurring during morning and evening rush hour traffic. Motor vehicle emissions consist mainly of NO, with only about 10% of primary emissions in the form of NO_2. The diurnal pattern of NO and NO_2 concentrations is also strongly influenced by the diurnal variation in the mixing layer height. Thus, during the morning rush hour when mixing layer heights are still low, traffic produces a peak in NO and NO_2 concentrations. As the mixing layer height increases during the day, dilution of emissions occurs. During the afternoon rush hour, mixing layer heights are at or are near their daily maximum values resulting in dilution of traffic emissions through a larger volume than in the morning. Starting near sunset, the mixing layer height drops and conversion of NO to NO_2 occurs without photolysis of NO_2 recycling NO.

The composite diurnal variability of NO_2 in selected urban areas with multiple sites (New York, NY; Atlanta, GA; Baton Rouge, LA; Chicago. IL; Houston, TX; Riverside, CA; and Los Angeles, CA) is shown in Figure AX3.3. Figure AX3.3 shows that lowest hourly median concentrations are typically found at around midday and that highest hourly median concentrations are found either in the early morning or in mid-evening. Median values range by about a factor of two from about 13 ppb to about 25 ppb. However, individual hourly concentrations can be considerably higher than these typical median values, and hourly NO_2 concentrations > 0.10 ppm can be found at any time of day.

AX3.2.2.2 Seasonal Variability in NO_2 Concentrations

Urban Sites

As might be expected from an atmospheric species that behaves essentially like a primary pollutant emitted from surface sources, there is strong seasonal variability in NO_x and NO_2 concentrations. Highest concentrations are found during winter, consistent with lowest mixing layer heights found during the year. Mean and peak concentrations in winter can be up to a factor of two larger than in the summer at several sites in Los Angeles County.

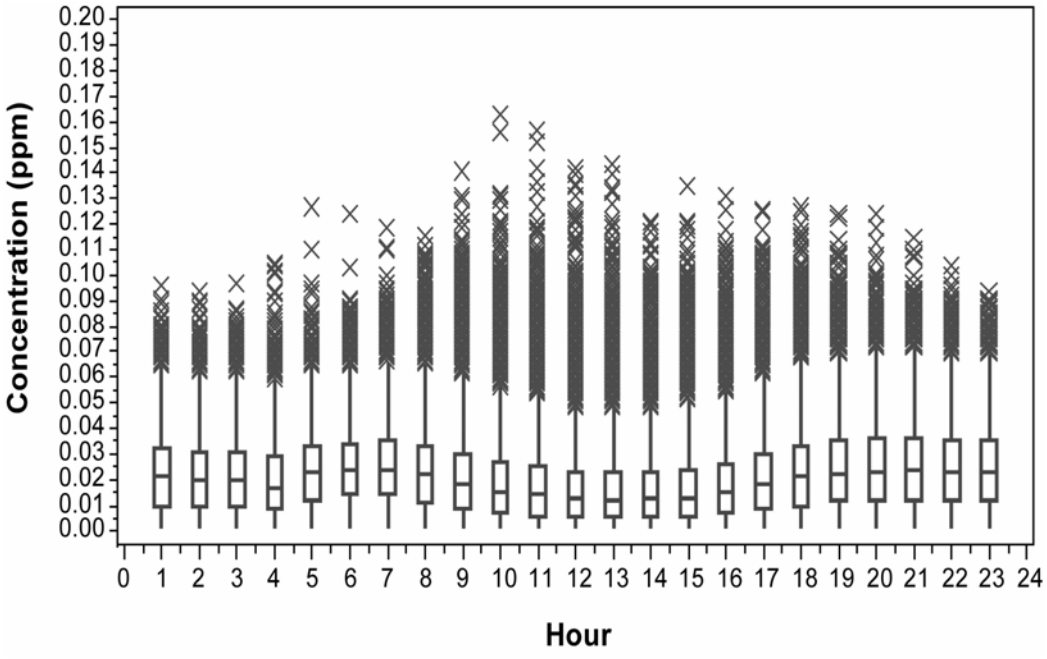

Figure AX3.3. Composite, diurnal variability in 1-h average NO_2 in urban areas. Values shown are averages from 2003 through 2005. Boxes define the interquartile range, and the whiskers the 5th and 95th percentile values. Asterisks denote individual values above the 95th percentile.

The month-to-month variability in NO_2 at individual sites in selected urban areas is illustrated in Figures AX3.4 to AX3.10. Seasonal patterns can be found at some sites but not in others. There appears to be a somewhat regular pattern for the southern cities with winter maxima and summer minima. Monthly maxima tend to be found from late winter to early spring in Chicago and New York with minima occurring from summer through the fall. However, in Los Angeles and Riverside, monthly maxima tend to occur from autumn through early winter with minima occurring from spring through early summer.

Regional Background Sites

Surface NO_x and NO_y data obtained in Shenandoah National Park, VA from 1988 to 1989 show wintertime maxima and summertime minima (Doddridge et al., 1991, 1992; Poulida et al., 1991). NO_x and NO_y data collected in Harvard Forest, MA from 1990 to 1993 show a similar seasonal pattern (Munger et al., 1996). In addition the within-season variability was found to be smaller in the summer than in the winter as shown in Table AX3.3.

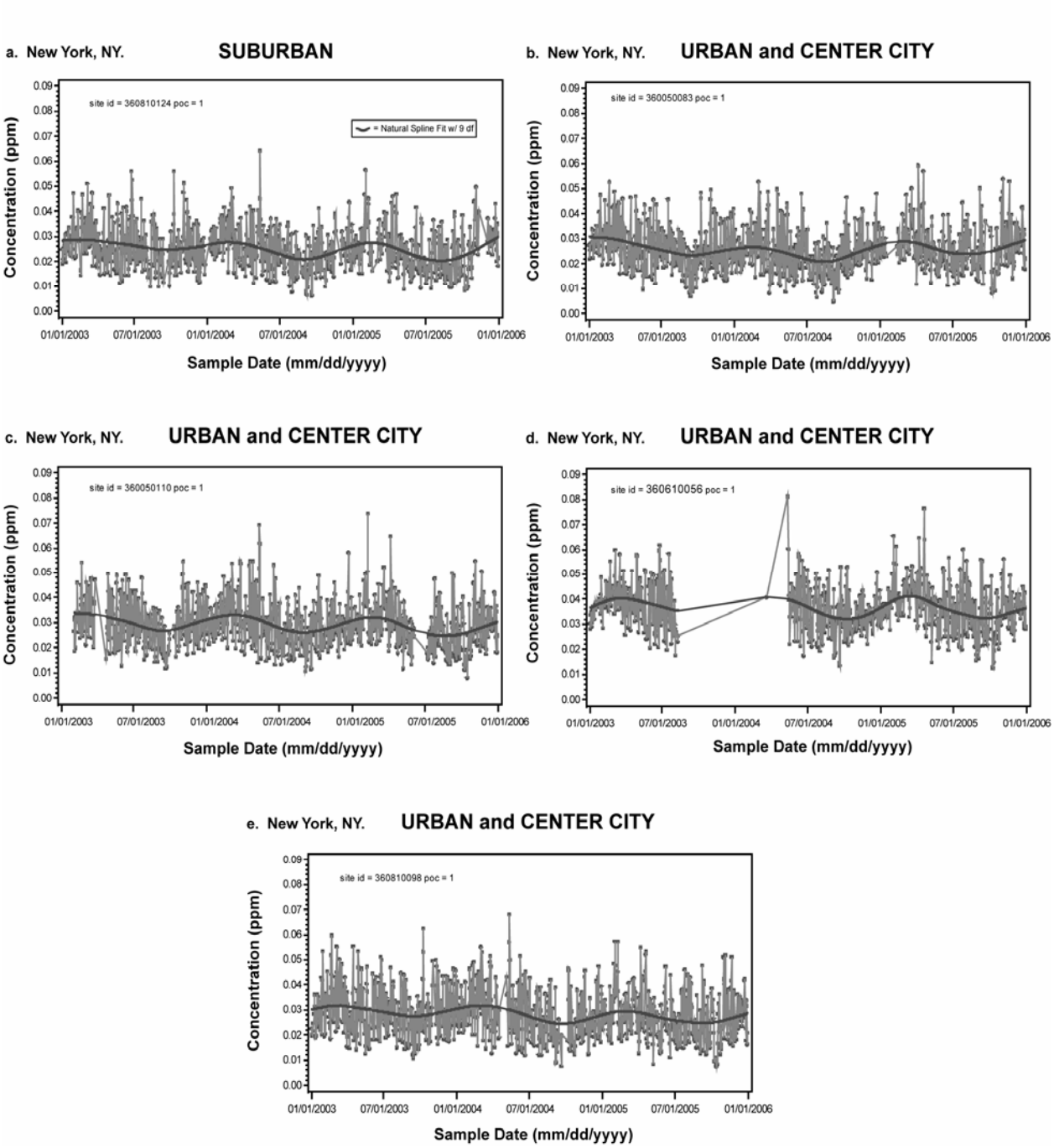

Figure AX3.4a-e. **Time series of 24-h average NO_2 concentrations at individual sites in New York City from 2003 through 2006. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).**

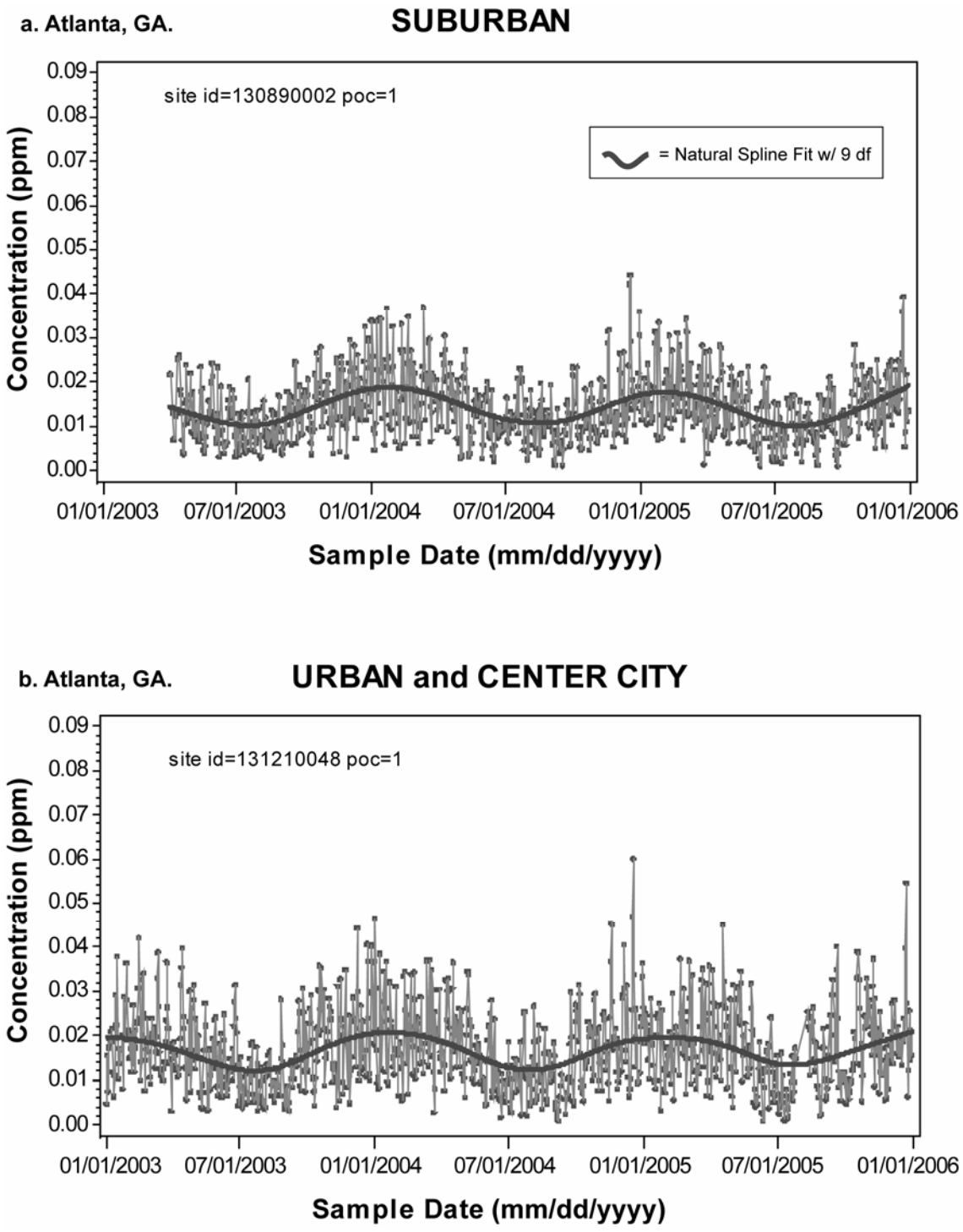

Figure AX3.5a-e. **Time series of 24-h average NO_2 concentrations at individual sites in Atlanta, GA from 2003 through 2005. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).**

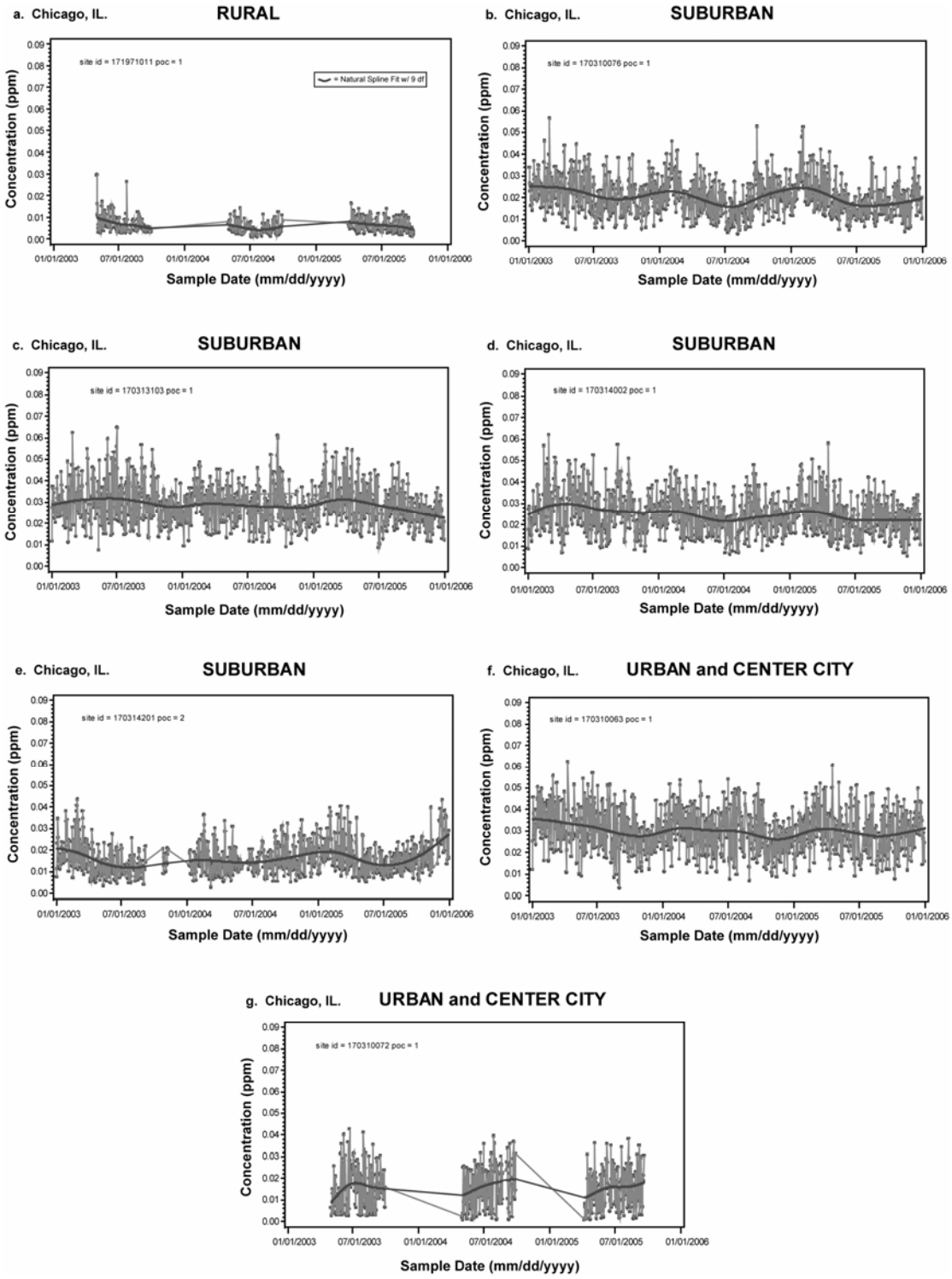

Figure AX3.6a-g. Time series of 24-h average NO_2 concentrations at individual sites in Chicago, IL from 2003 through 2005. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

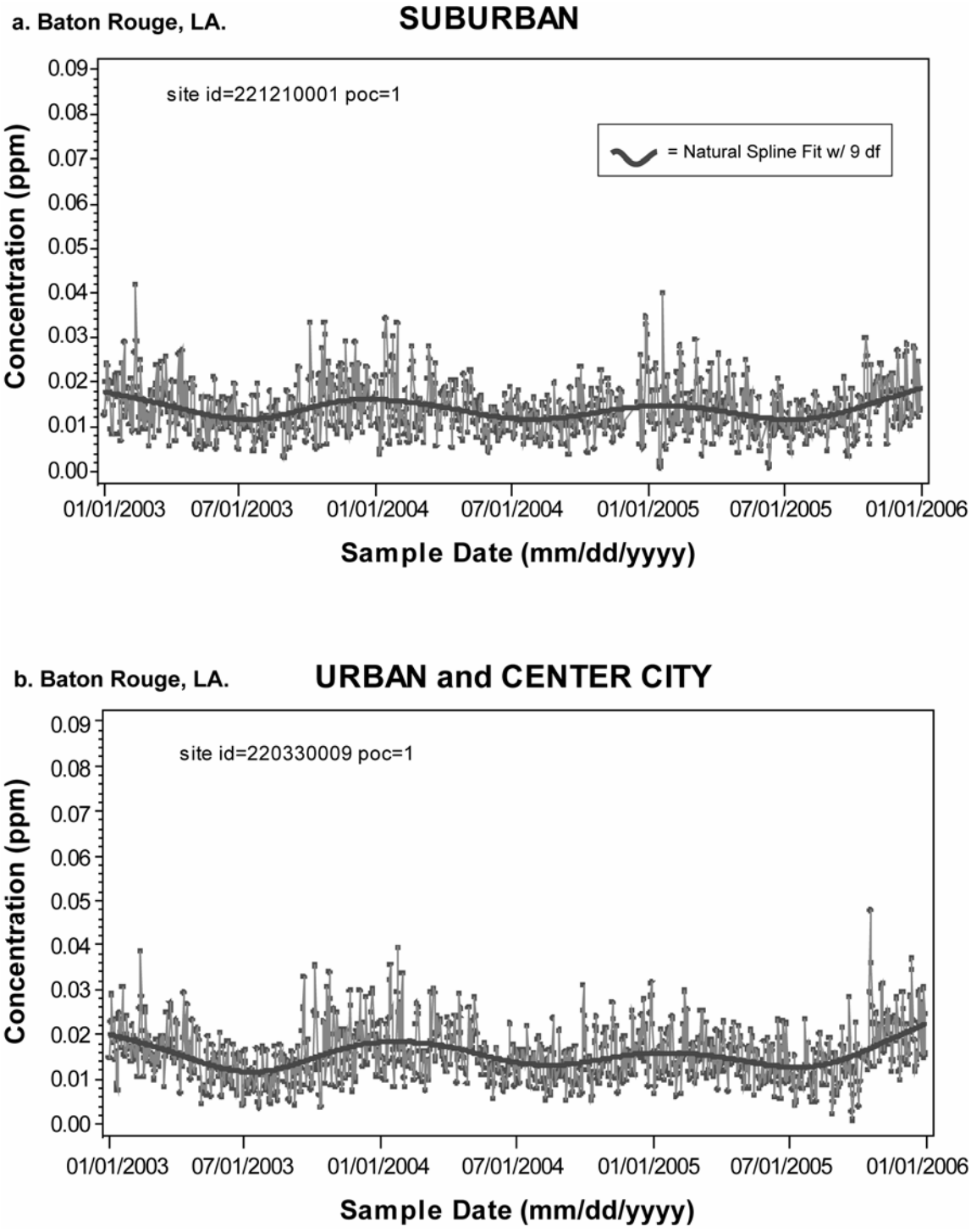

Figure AX3.7a-b. Time series of 24-h average NO_2 concentrations at individual sites in Baton Rouge, LA from 2003 through 2005. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

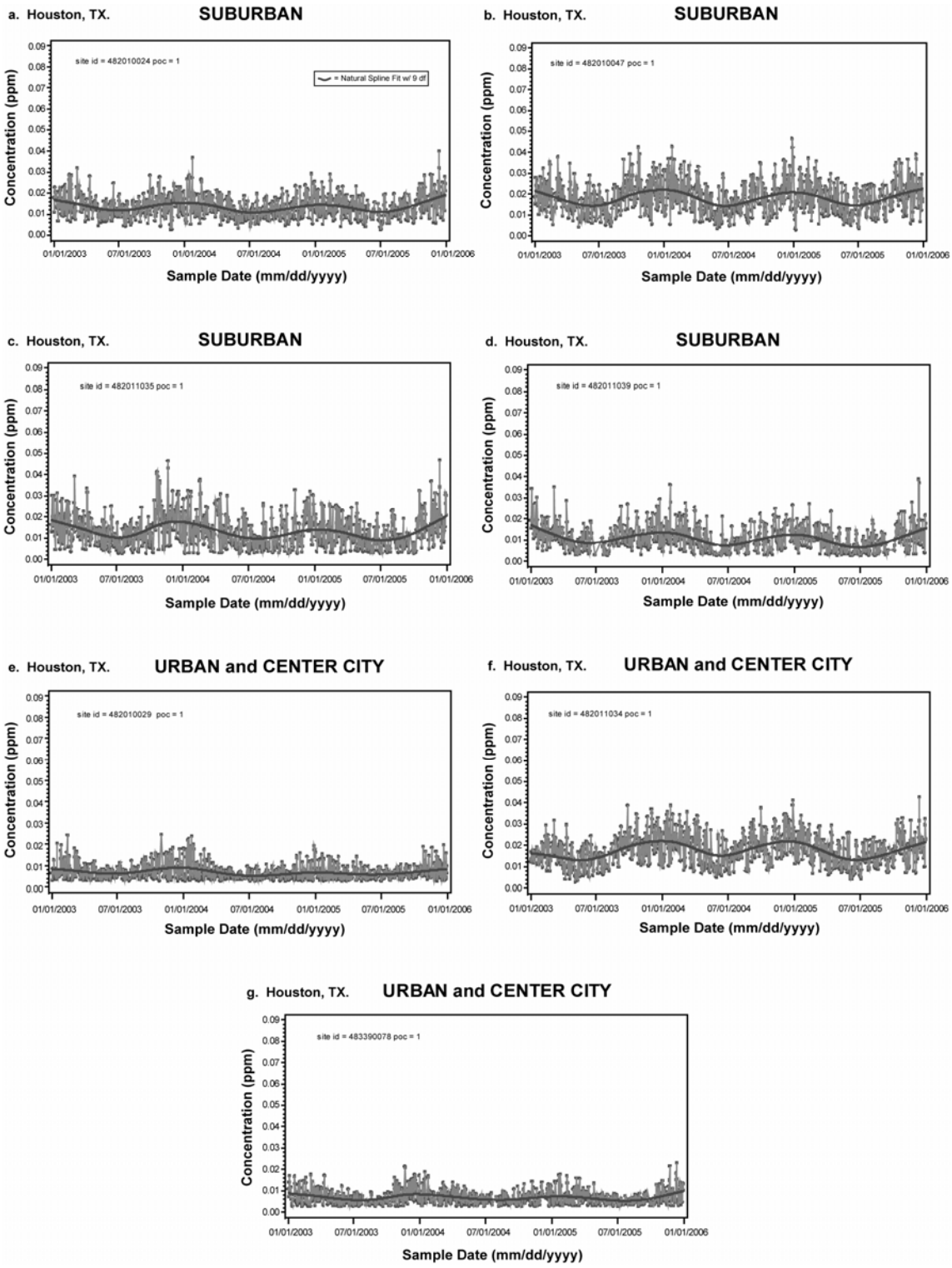

Figure AX3.8a-g. Time series of 24-h average NO_2 concentrations at individual sites in Houston, TX from 2003 through 2005. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

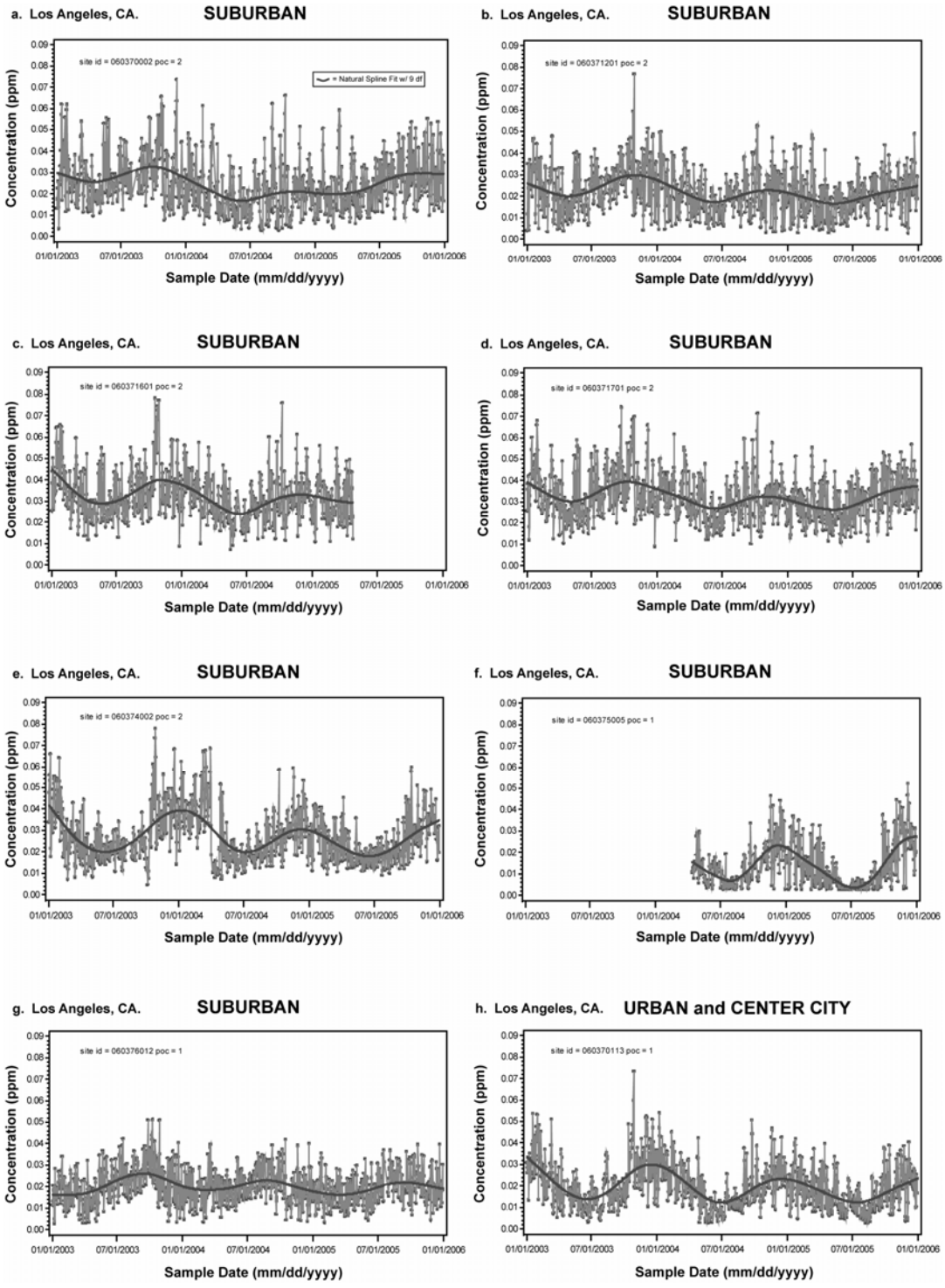

Figure AX3.9a-h. Time series of 24-h average NO_2 concentrations at individual sites in Los Angeles, CA from 2003 through 2005. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

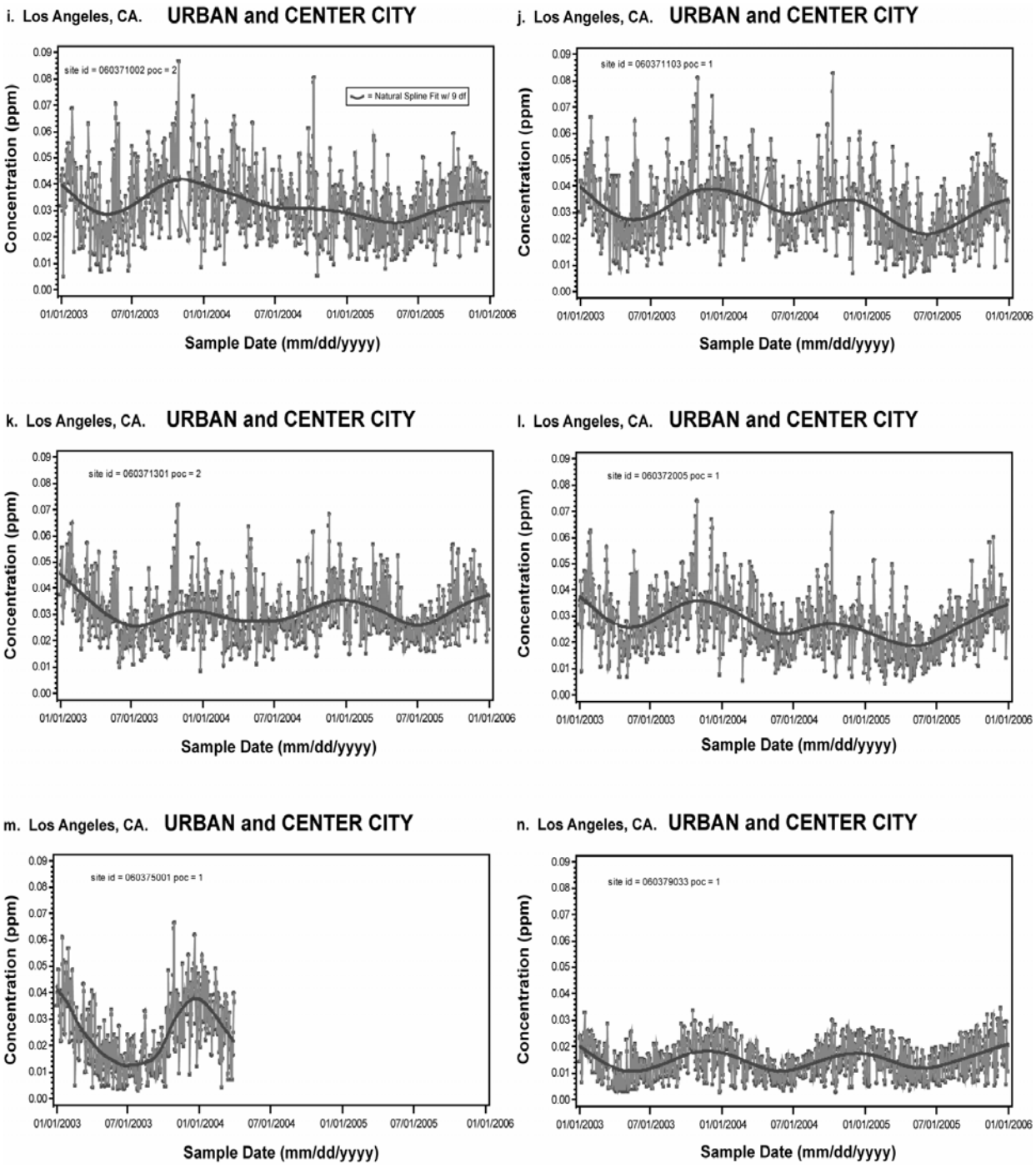

Figure AX3.9i-n. Time series of 24-h average NO_2 concentrations at individual sites in Los Angeles, CA from 2003 through 2006. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

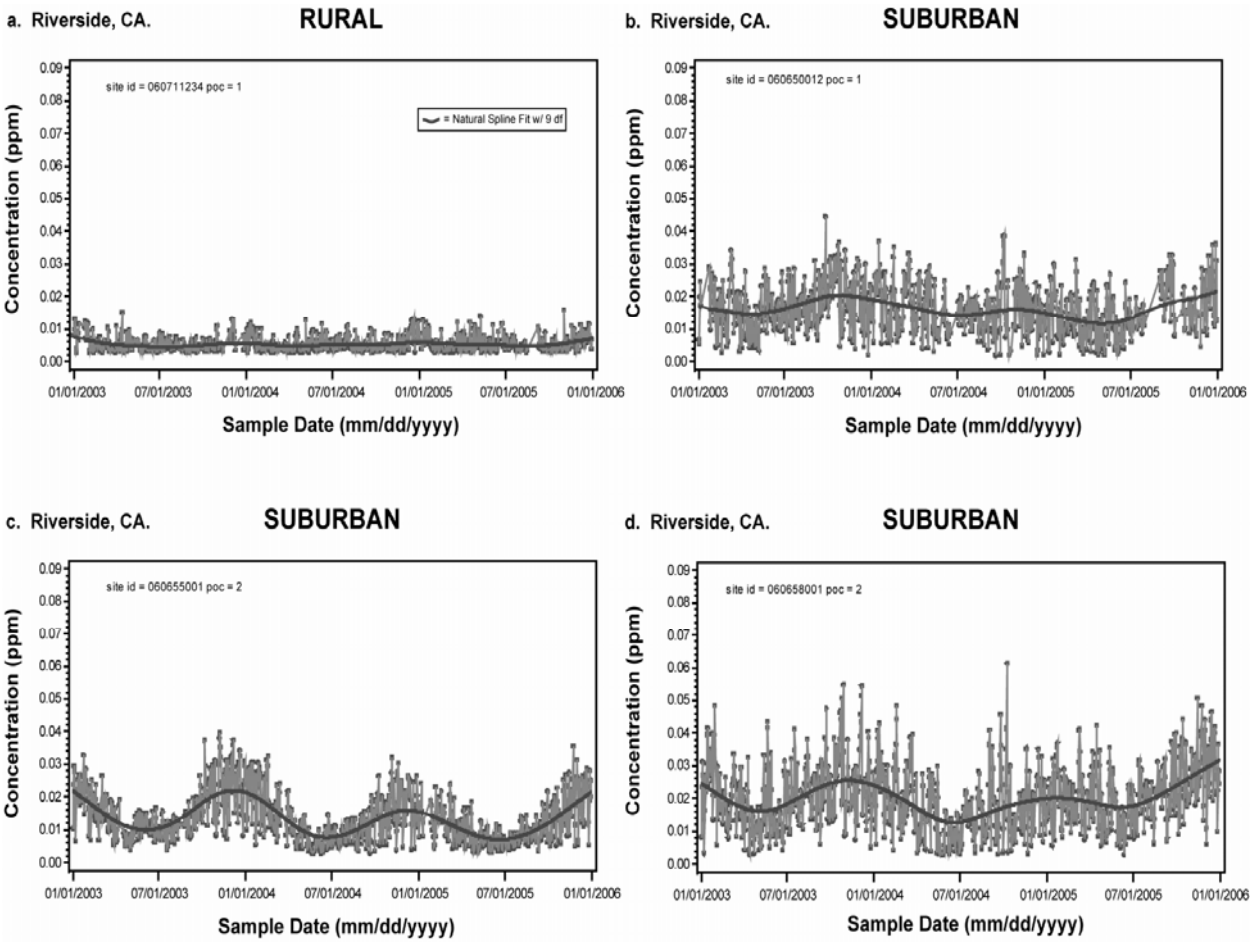

Figure AX3.10a-d. Time series of 24-h average NO_2 concentrations at individual sites in Riverside, CA from 2003 through 2006. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

AX3.2.2.3 Trends in NO_2 Concentrations

Figure AX3.11 shows the nationwide trend in annual mean NO_2 concentrations from 1983 to 2002. As can be seen from the figure, NO_2 concentrations have decreased by about 10% per decade. As can be seen from Figure AX3.12, most monitoring sites are located in either urban (49) or suburban (58) areas and comparatively few monitoring sites are located in rural areas (14). Figure AX3.12 also shows that decreases have been at least twice as large in urban and suburban areas than in rural areas and that NO_2 concentrations in urban and suburban areas are roughly twice those in rural areas. Note that a land use characterization of rural does not

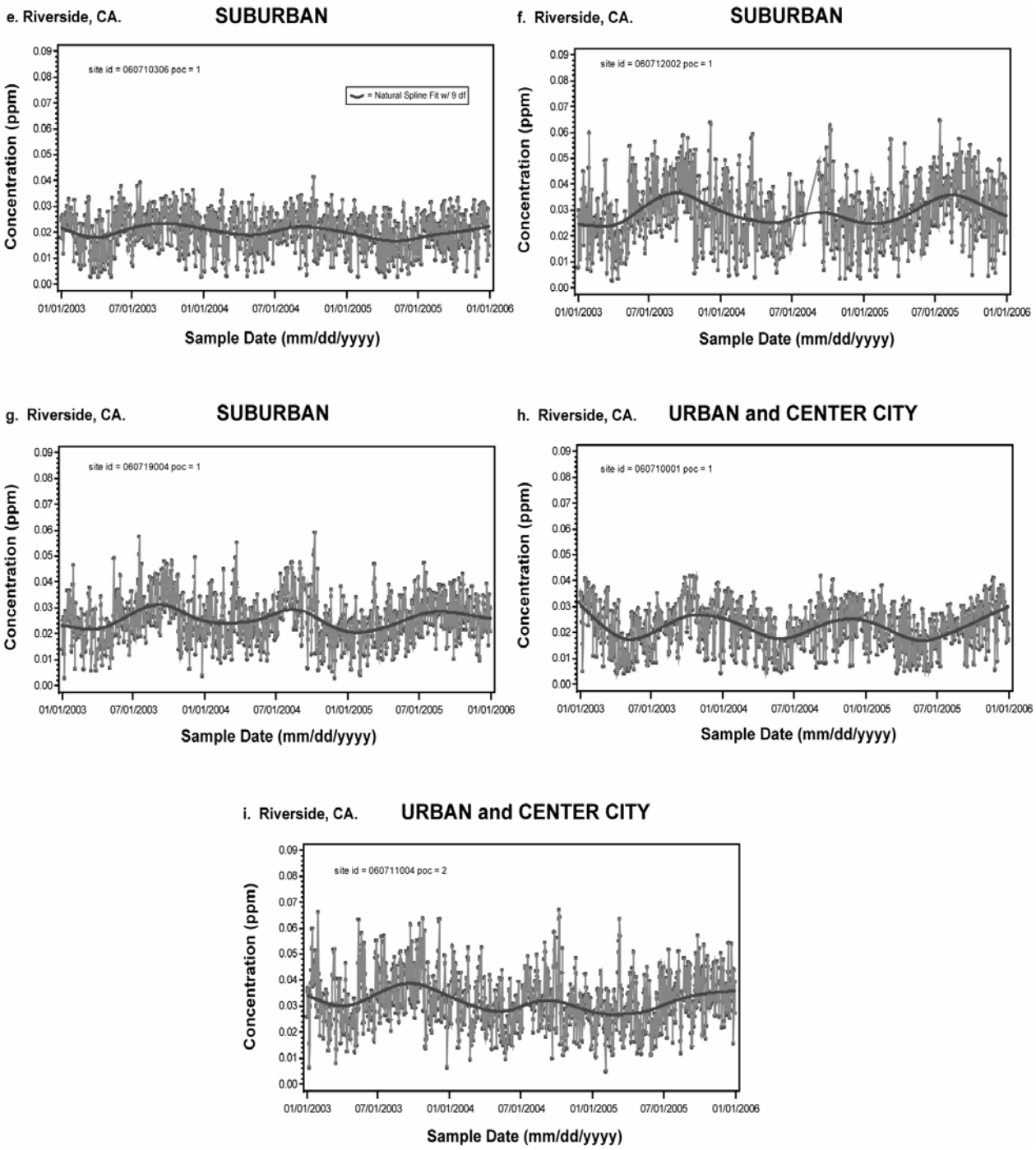

Figure AX3.10e-i. Time series of 24-h average NO_2 concentrations at individual sites in Riverside, CA from 2003 through 2006. A natural spline function (with 9 degrees of freedom) was fit and overlaid to the data (dark solid line).

imply that a site is free of local pollution influences, as evidenced by the still relatively high values at rural sites compared to those found in remote areas of the globe. Rural sites can be affected by nearby highways, power plants, and other sources.

In addition to the downward trend in annual mean concentrations of NO_2 shown in Figures AX3.11 and AX3.12, hourly maximum concentrations have also declined, as evidenced by a number of peak values above 250 ppb across the United States in 1988. In contrast only one hourly maximum concentration above 250 ppb was found in 2004 (however, this may have been a measurement artifact as it represented a one h spike that was many times the next highest concentration at this site), and all other values were less than about 150 ppb.

AX3.2.4 Relationships Between NO_2 and Other Pollutants

Determining the relationships between NO_2 and other pollutants is important for better understanding the findings of time-series epidemological studies relating NO_2 to mortality (e.g., Burnett et al., 2004). Correlations between NO_2 and CO, O_3, and $PM_{2.5}$ were calculated for monitoring sites in Los Angeles and Riverside, CA; Chicago, IL; Washington, D.C.; and New York City. Correlations were calculated using both hourly and 24-h average data with similar results. The ranges of Pearson correlation coefficients between 24-h average NO_2 and O_3, CO and $PM_{2.5}$ for 2000 through 2004 at monitoring sites in a few urban areas are shown in Table AX3.4. As can be seen from the table, correlations of NO_2 with O_3 range from negative to slightly positive; with CO they range from slightly negative to highly positive, and with $PM_{2.5}$ they range from slightly to moderately positive. However, it should be noted that these correlations are based on annual data from sites influenced by local sources. In general, there is a strong seasonal variation in the correlations, r, with lowest values of r between NO_2 and O_3 found in winter.

In order to understand the relations between atmospheric species as shown in Table AX3.4, an important distinction must be made between primary (directly emitted) species and secondary (photochemically produced) species. In general, it is more likely that primary species will be more highly correlated with each other, and that secondary species will be more highly correlated with each other. By contrast, primary and secondary species are less likely to be correlated with each other. Secondary reaction products tend to correlate with each other, but

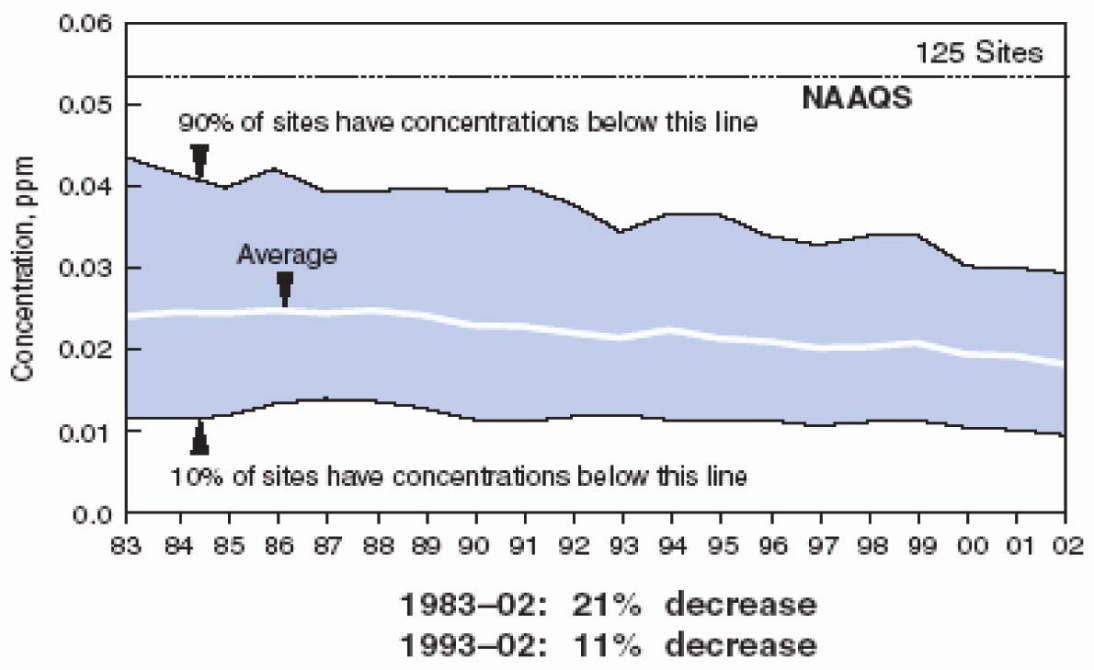

Figure AX3.11. Nationwide trends in annual mean NO_2 concentrations.

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

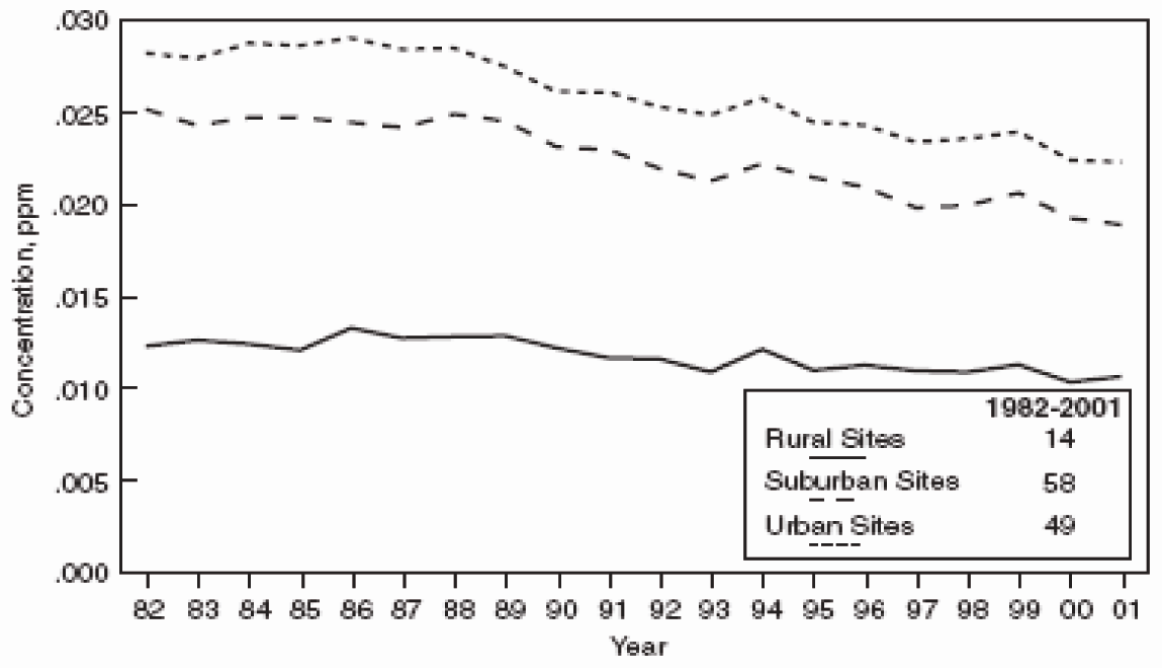

Figure AX3.12. Trends in annual mean NO_2 concentrations by site type.

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

there is considerable variation. Some species (e.g., O_3 and organic nitrates) are closely related photochemically and correlate with each other strongly.

Although NO_2 is produced mainly by the reaction of directly emitted NO with O_3 with a small contribution from direct emissions, in practice, it behaves like a primary species. The timescale for conversion of NO to NO_2 is relatively rapid (~1 or 2 min for O_3 = 40 ppb and ambient temperatures from 273 to 298 K), so NO and NO_2 ambient concentrations rapidly approach values determined by the photochemical steady state. The sum of NO and NO_2 (NO_x) behaves like a typical primary species, while NO and NO_2 reflect some additional complexity based on photochemical interconversion. Chemical interactions among O_3, NO and NO_2 have the effect of converting O_3 to NO_2 and vice versa, which can result in a significant negative correlation between O_3 and NO_2.

Most CO in urban air is emitted from motor vehicles and so is primary in origin. O_3 is a secondary pollutant. Figures AX3.13a-d show seasonal plots of correlations between NO_2 and O_3 versus correlations between NO_2 and CO. As can be seen from the figures, NO_2 is positively correlated with CO during all seasons at all sites. However, the sign of the correlation of NO_2 with O_3 varies with season, ranging from negative during winter to slightly positive during summer. There are at least two main factors contributing to the observed seasonal behavior. O_3 and radicals correlated with it tend to be higher during the summer, thereby tending to increase the NO_2 to NO ratio according to the expression below (Equation AX3-2).

$$\frac{NO_2}{NO} = \frac{k_1(O_3) + k_2(HO_2) + k_3(RO_2)}{J(NO_2)} \qquad \text{(AX3-2)}$$

NO_z compounds formed from the oxidation of NO_x are also expected to be correlated with O_3 and increased photochemical activity. Because of interference of NO_z compounds with the measurement of NO_2 by conventional chemiluminescent monitors, they may also tend to increase the correlation of NO_2 with O_3 during the warmer months. However, there is not enough information on the seasonal behavior in their concentrations to quantify the contribution of NO_z compounds.

Relationships between O_3, NO, and NO_2 are shown in Figures AX3.14 and AX3.15. Figure AX3.14 shows daylight average concentrations based on data collected from November 1998 and 1999 at several sites in the United Kingdom representing a wide range of pollution

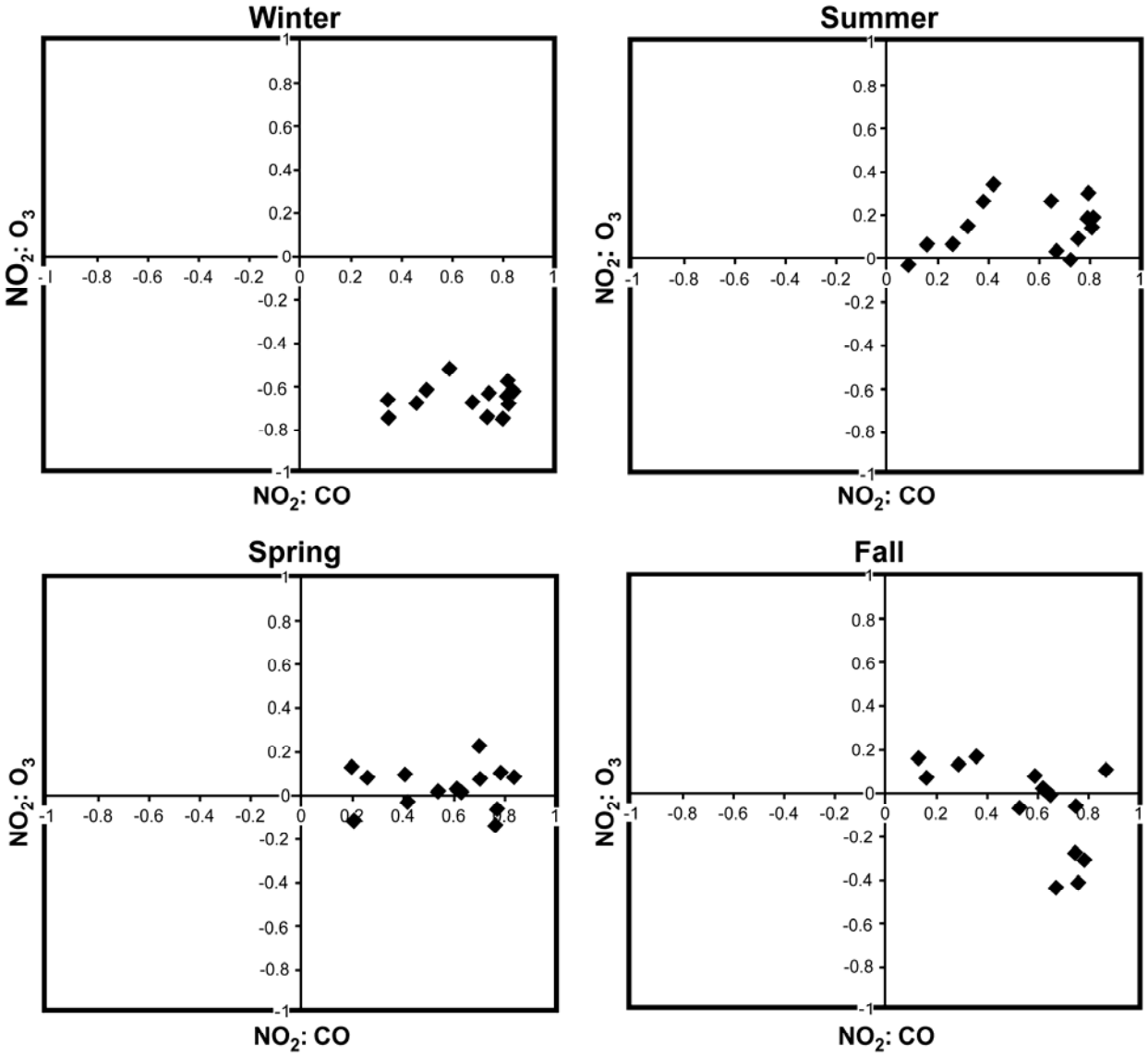

Figure AX3.13a-d. Correlations of NO_2 to O_3 vs. correlations of NO_2 to CO for Los Angeles, CA (2001-2005).

conditions (open symbols). The solid lines represent calculations of photostationary state values subject to the constraint that $O_x = 31.1 + 0.104(NO_x)$, where $O_x = O_3 + NO_2$. Note that O_x is defined in the UK AQG report as oxidant, as used in this document, and in the latest AQCD for Ozone and other Photochemical Oxidants (U.S. Environmental Protection Agency, 2006) it is taken to refer to "odd oxygen" as defined in Section 2.2. The reason is that oxidants also include PANs, peroxides, and reactive oxygen species in particles etc., in addition to O_3 and NO_2. The

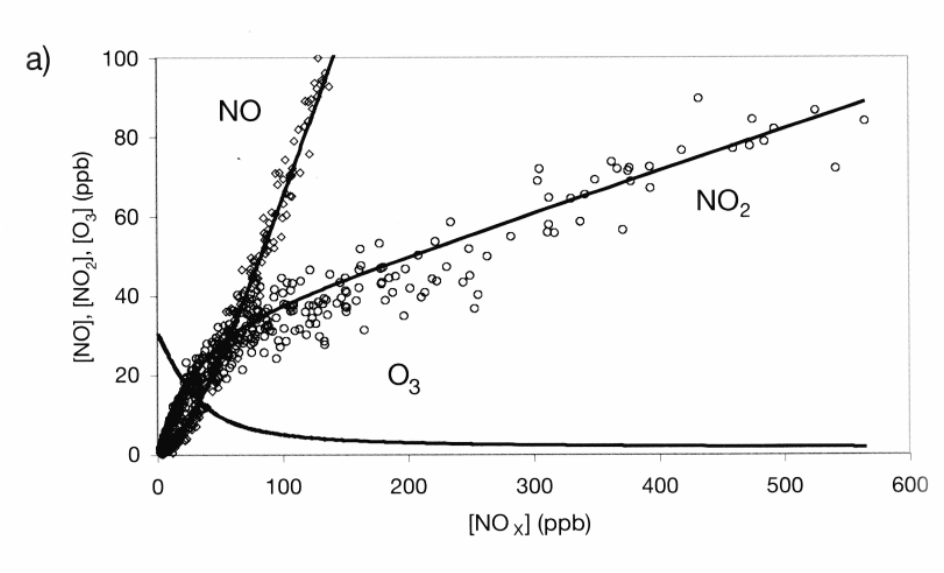

Figure AX3.14. Relationship between O_3, NO, and NO_2 as a function of NO_x concentration. Open circles represent data collected at a number of sites in the United Kingdom. Lines represent calculated relationships based on photostationary state.

Source: Clapp and Jenkin (2001).

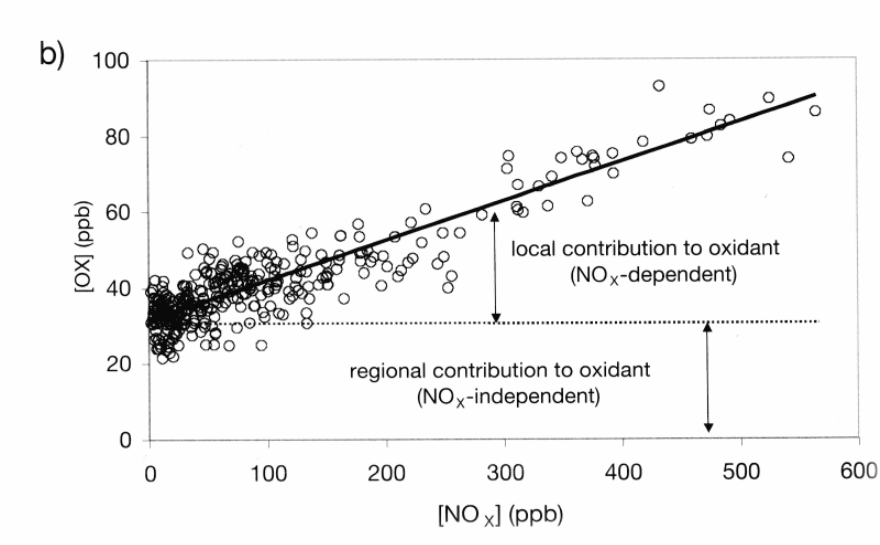

Figure AX3.15. Variation of odd oxygen (= O_3 + NO_2) with NO_x. The figure shows the "regional" and the "local" contributions. Note that O_x refers to odd oxygen in the document and the latest O_3 AQCD.

Source: Clapp and Jenkin (2001).

intercept of O_3 with the y-axis at about 30 ppb is representative of background values of O_3 in the UK. The figure shows how O_3 decreases with increasing NO and NO_x. NO constitutes more than about 90% of NO_x at high values of NO_x as available O_3 is titrated away.

Figure AX3.15 shows how the concentration of O_x (=O_3 + NO_2) varies with that of NO_x. As in Figure AX3.14, O_3 intercepts the y-axis at about 30 ppb, corresponding to background O_x which is composed almost exclusively of O_3. O_x increases in a linear fashion, as given by the regression relation above, as NO_x increases. This relationship results from the emissions of NO_2 (an oxidant and a component of odd oxygen) varying linearly with emissions of NO_x, especially after NO has reacted with O_3 to form NO_2 as shown in Figure AX3.14. Thus the concentration of O_x (and not O_3, as is often stated) can be taken to be the sum of regional and local contributions.

Figure AX3.15 shows that primary emissions from motor vehicles are major sources of oxidant in the form of NO_2, as evidenced by the high values of O_x at elevated NO_x.

AX3.2.5 Abundance of NO_y Species

Data for individual NO_y species are much less abundant than for either oxides of nitrogen or for total NO_y. Data for NO_y species are collected typically as part of research field studies, e.g., the Southern Oxidant Study (SOS), Texas Air Quality Study (TexAQS I and TexAQS II) in the United States. So this information is simply not available for a large number of areas in the United States.

PANs

Organic nitrates consist of PAN, a number of higher-order species with photochemistry similar to PAN (e.g., PPN), and species such as alkyl nitrates with somewhat different photochemistry. These species are produced by a photochemical process very similar to that of O_3. Photochemical production is initiated by the reaction of primary and secondary VOCs with OH radicals, the resulting organic radicals subsequently react with NO_2 (producing PAN and analogous species) or with NO (producing alkyl nitrates). The same sequence (with organic radicals reacting with NO) leads to the formation of O_3.

In addition, at warm temperatures, the concentration of PAN forms a photochemical steady state with its radical precursors on a timescale of roughly 30 min. This steady state value increases with the ambient concentration of O_3 (Sillman et al., 1990). Ozone and PAN may

show different seasonal cycles, because they are affected differently by temperature. Ambient O_3 increases with temperature, driven in part by the photochemistry of PAN (see description in Chapter 2). The atmospheric lifetime of PAN decreases rapidly with increasing temperature due to thermal decomposition. Based on the above, the ratio of O_3 to PAN is expected to show seasonal changes, with highest ratios in summer, although there is no evidence from measurements. Measured ambient concentrations (Figures AX3.16a-d) show a strong nonlinear association between O_3 and PAN, and between O_3 and other organic nitrates (Pippin et al., 2001; Roberts et al., 1998). Moreover, the uncertainty in the relationship between O_3 and PAN grows as the level of PAN increases. Individual primary VOCs are generally highly correlated with each other and with NO_x (Figure AX3.17).

Measurements and models show that PAN in the United States includes major contributions from both anthropogenic and biogenic VOC precursors (Horowitz et al., 1998; Roberts et al., 1998). Measurements in Nashville during the 1999 summertime Southern Oxidants Study (SOS) showed PPN and MPAN amounting to 14% and 25% of PANs, respectively (Roberts et al., 2002). Measurements during the TexAQS 2000 study in Houston indicated PAN concentrations of up to 6.5 ppbv (Roberts et al., 2003). PAN measurements in southern California during the SCOS97-NARSTO study indicated peak concentrations of 5-10 ppbv, which can be contrasted to values of 60-70 ppbv measured back in 1960 (Grosjean, 2003). Vertical profiles measured from aircraft over the United States and off the Pacific coasts typically show PAN concentrations above the boundary layer of only a few hundred pptv, although there are significant enhancements associated with long-range transport of pollution plumes from Asia (Kotchenruther et al., 2001a; Roberts et al., 2004).

Observed ratios of PAN to NO_2 as a function of NO_x at a site at Silwood Park, Ascot, Berkshire, UK are shown in Figure AX3.18 United Kingdom Air Quality Expert Group (U.K. AQEG, 2004). As can be seen there is a very strong inverse relation between the ratio and the NO_x concentration, indicating photochemical oxidation of NO_x has occurred in aged air masses and that PAN can make a significant contribution to measurements of NO_2 especially at low levels of NO_2 (cf. Section 2-8). It should be noted that these ratios will likely differ from those found in the U.S. because of differences in the composition of precursor emissions, the higher solar zenith angles found in the UK compared to the U.S., and different climactic conditions.

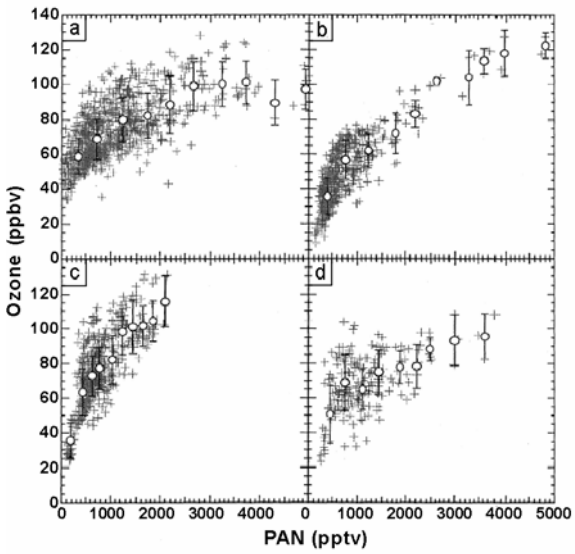

Figure AX3.16a-d. Measured O_3 (ppbv) versus PAN (pptv) in Tennessee, including (a) aircraft measurements, and (b, c, and d) suburban sites near Nashville.

Source: Roberts et al. (1998).

Nevertheless, these results indicate the potential importance of interference from NO_y compounds in measurements of NO_2.

HONO

The ratio of HONO to NO_2 as a function of NO_x measured at a curbside site in a street canyon in London, UK is shown in Figure AX3.19 (U.K. AQEG, 2004). The ratio is highly variable, ranging from about 0.01 to 0.1, with a mean ~0.05. As NO_2 constitutes several percent of motor vehicle emissions of NO_x, the above implies that emissions of HONO represent a few tenths of a percent of mobile NO_x emissions. A similar range of ratios have been observed at other urban sites in the United Kingdom (Lammel and Cape, 1996). The ratios of HONO to NO2 shown in Figure AX3.19 indicate that HONO can make a measurable contribution to measurements of NO_2 (cf. Section 2-8). However, similar arguments about extrapolating the use of UK data to the U.S. can be made for HONO as for PAN.

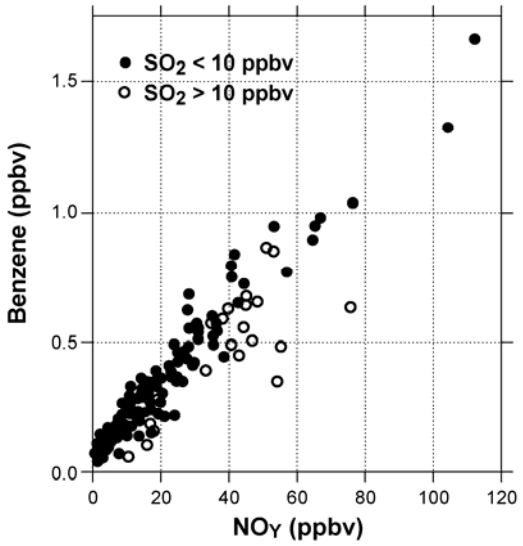

Figure AX3.17. Relationship between benzene and NO_y at a measurement site in Boulder, CO. Instances with SO_2 >10 ppb are identified separately (open circles), because these may reflect different emission sources.

Source: Goldan et al. (1995).

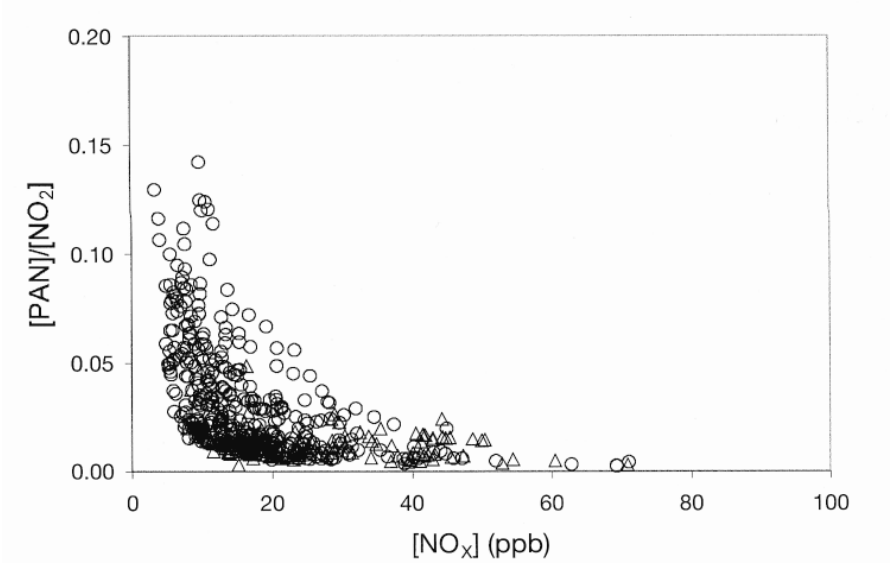

Figure AX3.18. Ratios of PAN to NO_2 observed at Silwood Park, Ascot, Berkshire, U.K. from July 24 to August 12 1999. Each data point represents a measurement averaged over 30 minutes.

Source: UK AQEG (2004).

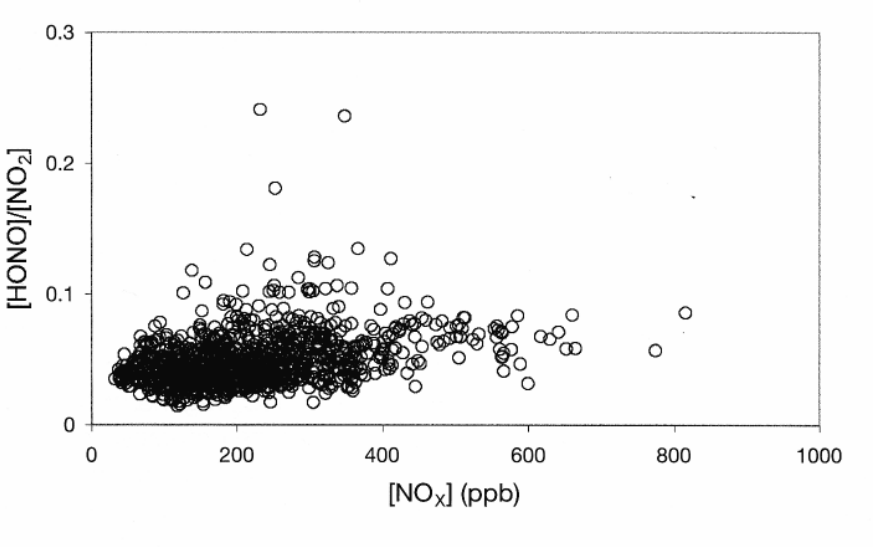

Figure AX3.19. Ratios of HONO to NO_2 observed in a street canyon (Marylebone Road) in London, U.K. from 11 a.m. to midnight during October 1999. Data points reflect 15-min average concentrations of HONO and NO_2.

Source: UK AQEG (2004).

HNO_3 and NO_3

Elevated O_3 is generally accompanied by elevated HNO_3, although the correlation is not as strong as between O_3 and organic nitrates. Ozone is often associated with HNO_3, because they have the same precursor (NO_x). However, HNO_3 can be produced in significant quantities in winter, even when O_3 is low. The ratio between O_3 and HNO_3 also shows great variation in air pollution events, with NO_x-saturated environments having much lower ratios of O_3 to HNO_3 (Ryerson et al., 2001). Aerosol nitrate is formed primarily by the combination of nitrate (supplied by HNO_3) with ammonia, and may be limited by the availability of either nitrate or ammonia. Nitrate is expected to correlate loosely with O_3 (see above), whereas ammonia is not expected to correlate with O_3.

Concentrations of particulate nitrate measured as part of the Environmental Protection Agency's speciation network at several locations are shown in Figure AX3.20. Concentrations shown are annual averages for 2003. Also shown are the estimated contributions from regional and local sources. A concentration of 1 µg/m^3 corresponds to ~0.40 ppb equivalent gas phase concentration for NO_3-. Thus, annual average particulate nitrate can account for several ppb of

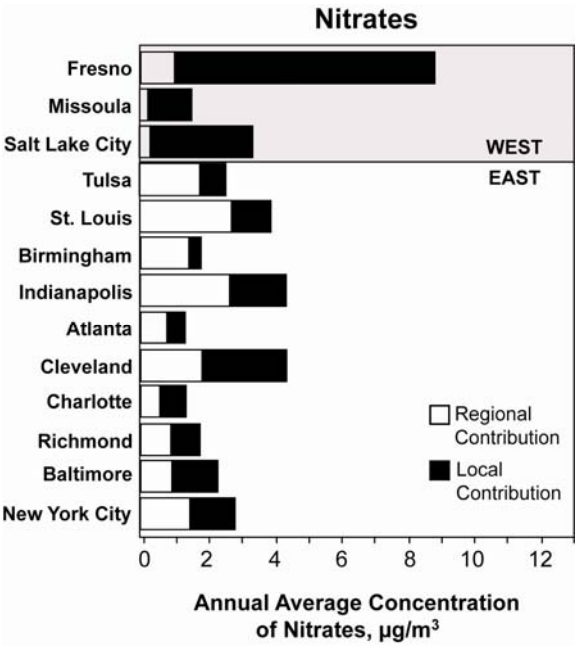

Figure AX3.20. Concentrations of particulate nitrate measures as part of the Environmental Protection Agency PA's speciation network. 1 µg/m^3 ~0.45 ppb equivalent gas phase concentration for NO_3^-. (Note: Regional concentrations are derived from the rural IMPROVE monitoring network, http://vista.cira.colostate.edu/improve.

Source: U.S. Environmental Protection Agency (2004).

NO_y, with the higher values in the West. There is a strong seasonal variation, which is especially pronounced in western areas where there is extensive wood burning in the winter resulting in a larger fractional contribution of local sources. Areas in the East where there are topographic barriers might be expected to show higher fractional contributions from local sources than other eastern areas that are influenced by regionally dispersed sources.

However, depending on the acidity of the particles, which in turn depends strongly on their sulfate and ammonium contents, higher nitrate concentrations could be found in coarse mode particles $PM_{10-2.5}$ than in $PM_{2.5}$ samples. The average nitrate content of $PM_{2.5}$ and PM_{10} is typically about a percent in the eastern United States; and 15.7% and 4.5% in the western United States (U.S. Environmental Protection Agency, 1996). These values suggest that most of the nitrate was in the $PM_{2.5}$ size fraction in the studies conducted in the western United States, but nitrate in the studies in the eastern United States was mainly in the $PM_{10-2.5}$ size fraction.

Nitro-PAHs

Nitro-PAHs (NPAHs) are widespread and found even in high altitude, relatively unpolluted environments (Schauer et al., 2004) but there are differences in composition and concentration profiles both within and between sites (rural vs. urban) as well as between and within urban areas (Albinet et al., 2006; Söderström et al., 2005; Naumova et al., 2002, 2003), with some differences in relative abundances of nitro- and oxo-PAHs also reported. Source attribution has remained largely qualitative with respect to concentrations or mutagenicity (Eide et al., 2002). The spatial and temporal concentration pattern for the NPAHs may differ from that of the parent compounds (PAHs) because concentrations of the latter are dominated by direct emission from local combustion sources. These emissions results in higher concentrations during atmospheric conditions more typical of wintertime when mixing heights tend to be low. The concentrations of secondary NPAHs are elevated under conditions that favor hydroxyl and nitrate radical formation, i.e., during conditions more typical of summertime, and are enhanced downwind of areas of high emission density of parent PAHs and show diurnal variation (Fraser et al., 1998; Reisen and Arey, 2005; Kameda et al., 2004). Nitro-napthalene concentrations in Los Angeles, CA varied between about 0.15 to almost 0.30 ng/m^3 compared to 760 to 1500 ng/m^3 for napthalene. Corresponding values for Riverside, CA were 0.012 to more than 0.30 ng/m^3 for nitro-napthalene and 100 to 500 ng/m^3 for napthalene. Nitro-pyrene concentrations in LA varied between approximately 0.020 to 0.060 ng/m^3 compared to 3.3 to 6.9 ng/m^3 pyrene, whereas corresponding values for Riverside were 0.012 to 0.025 ng/m^3 and 0.9 to 2.7 ng/m^3.

AX3.3 METHODS FOR MEASURING PERSONAL AND INDOOR NO_2 CONCENTRATIONS

AX3.3.1 Issues in Measuring Personal/Indoor NO_2

Background

Nitrogen dioxide, a criteria air pollutant, has been sampled in ambient and indoor air using active pumped systems both for continuous monitoring and collection onto adsorbents, and by diffusive samplers of various designs, including badges and tubes. Nitrogen dioxide concentrations in personal air have been typically measured using diffusive samplers because

they are: (1) small in size and light-weight, (2) unobtrusive and thus more readily used by study participants, (3) comparatively easier to use and handle in field studies because they do not require power (e.g., battery or extra electrical sources), (4) cost-effective, and (5) usable not only for residential indoor and outdoor air sampling but also personal monitoring. However, diffusive samplers usually have lower equivalent sampling rates than active methods and so require relatively long sampling times (24 h or longer). Consequently, diffusive samplers including those used for NO_2 monitoring provide integrated but not short-term concentration measurements.

Both active and passive sampling methods can collect other gas-phase nitrogen oxide species. However, semivolatile nitrogen oxide compounds require separation of the gas- and particle-bound phases. This selective separation of gases from gas-particle matrices is commonly done by means of diffusion denuders (Vogel, 2005), an approach also useful for measuring other gas phase airborne contaminants such as SO_2 (Rosman et al., 2001). Application of denuder sampling to personal exposure or indoor air monitoring has been relatively limited.

Active air sampling with a pump can collect larger volumes of air and thus detect the lower concentrations found in community environments within relatively short time periods. Automated active sampling methods have been the preferred method used to monitor NO_2 continuously at ambient sites for environmental regulation compliance purposes. However, practical considerations impede the use of these continuous monitors in residential air and exposure monitoring studies. Small, low flow active samplers using battery-operated pumps have been used instead, however, there are only a few such studies.

The first passive sampling devices for NO_2 were intended for occupational exposure monitoring, but were later adapted for environmental monitoring purposes. Since this sampler, the Palmes tubes (Palmes et al., 1976), was first developed, other tube, badge-type (Yanagisawa and Nishimura, 1982) and radial (Cocheo et al., 1996) diffusive samplers have been employed as monitors in exposure studies worldwide. The theories behind and applications of Palmes Tubes and Yanagisawa badges have been described in the last AQCD for Oxides of Nitrogen (U.S. Environmental Protection Agency, 1993). There are currently several commercially available samplers (e.g., Ogawa, Radiello®, Analyst™) which are modifications of the original Palmes tube design. Most modifications are directed at reducing effects related to meteorological

conditions (e.g., insufficient or too high a wind speed, humidity, temperature), increasing the sampling uptake rate, and improving analytical sensitivity.

Active (Pumped) Sampling

Nitrogen dioxide measurement by active pumping systems as part of continuous monitors has been widely employed for ambient air monitoring as these instruments require relatively little maintenance; however they have been used less frequently for indoor sampling. Devices needing a pump to draw air can measure average concentrations of pollutants over short time periods, but are not generally suitable for measuring personal exposures because they are heavy and large. Some exposure studies employed this approach for active sampling with stationary chemiluminescent analyzers or portable monitors to measure nitrogen dioxide levels in residential indoor air (Mourgeon et al., 1997; Levesque et al., 2000; Chau et al., 2002). Recently, Staimer and his colleagues (2005) evaluated a miniaturized active sampler, suitable for personal exposure monitoring, to estimate the daily exposure of pediatric asthmatics to nitrogen dioxide, and reported that this small active sampling system is useful for this purpose in environmental exposure epidemiology studies where daily measurements are desired.

Passive (Diffusive) Sampling

Passive samplers are based on the well known diffusion principle described by Fick's law (Krupa and Legge, 2000). A convenient formulation of this law that can be easily related to sampler design considerations is:

$$J = D(A/L)(C_{air}\text{-}C_{sor}) \qquad \text{(AX3-3)}$$

where:

J = flux (mg/s)

D = diffusion coefficient in air (cm^2/s)

A = diffusion cross-sectional area of the sampler (cm^2)

L = diffusion path length from the inlet to sorbent (cm),

C_{air} = concentration of analyte in air (mg/cm^3)

C_{sor} = concentration of analyte at the sorbent (mg/cm^3)

The term D(A/L) can be related to the uptake or sampling rate (cm^3/s) which is conceptually analogous to the sampling rate in an active monitor. Once the amount of analyte in the passive sampler sorbent is determined, the concentration in air (C_{air}) can be calculated as:

$$Concentration(mg/cm^3) = M(mg)/D(A/L)(cm^3/s)/t(sec) \quad \text{(AX3-4)}$$

where:

M = mass of analyte collected in the sorbent

t = sampling time

Fick's law strictly applies only under ideal, steady state conditions assuming that the sorbent is a perfect sink. However, there can be deviations between the theoretical sampling rate for a given analyte and the actual rate depending on sampling conditions. It is also clear that sampling rate can be optimized by modifying the geometry of the diffusive sampler, either by reducing L, increasing A or a suitable combination. However, the impact of deviations from ideality on actual sampling rate due to geometry also poses a limit to the extent of possible modifications. Thus, passive samplers, either diffusive or permeation, are prepared as tubes or badges. These two main designs are the basis for all further modifications which, as indicated above, have been made in order to improve efficiency, reduce sensitivity to wind turbulence of the samplers, and to simplify analyte desorption. Tube-type samplers are characterized by a long, axial diffusion length, and a low cross-sectional area; this results in relatively low sampling rates (Namiesnik et al., 2005). Badge-type samplers have a shorter diffusion path length and a greater cross-sectional area which results in uptake rates that are typically higher than diffusion tubes (Namiesnik et al., 2005) but the sampling rate may be more variable because it is more affected by turbulence. Physical characteristics of these two fundamental passive sampler types, tube-type and badge-type, are summarized and provided in Table AX3.5. Performance characteristics are presented in Table AX3.6.

The sorbent can be either physically sorptive or chemisorptive; passive samplers for NO_2 are chemisorptive, that is, a reagent coated on a support (e.g., metal mesh, filter) reacts with the NO_2. The sorbent is extracted and analyzed for one or more reactive derivatives; the mass of NO_2 collected is derived from the concentration of the derivative(s) based on the stoichiometry of the reaction. Thus, an additional approach to reducing detection limits associated with passive

samplers is to modify the chemisorptive reaction and the extraction and analysis methods to increase analytical sensitivity. However, although chemisorption is less prone to the back diffusion phenomenon of sorptive-only methods, analyte losses could occur due to interferences from other pollutants that also react with the sorbent or the derivatives. The most commonly used NO_2 passive samplers rely on the classical reaction with triethanolamine (TEA). TEA requires hydration for quantitative NO_2 sampling (i.e., 1:1 conversion to nitrite) and the reaction products have been subject to a number of investigations and several have been reported, including TEA-nitrate and nitrite, triethanolammonium nitrate, nitrosodiethanolamine, and triethanolamine N-oxide (Glasius et al., 1999). Known interferences include HONO, PAN, and nitric acid (Gair et al., 1991.).

The tube-type passive samplers (Palmes tubes) require week-long sampling periods and have been extensively used for residential indoor/outdoor measurements, mostly for exploring the relationship between indoor and outdoor levels (Cyrys et al., 2000; Raw et al., 2004; Simoni et al., 2004; Janssen et al., 2001). Passive diffusion tubes have also been widely used for measurements of NO_2 in ambient air (Gonzales et al., 2005; Gauderman et al., 2005; Da Silva et al., 2006; Lewne et al., 2004; Stevenson et al., 2001; Glasius et al., 1999). Personal exposure studies have also been conducted using the Palmes tubes (Mukala et al., 1996; Kousa et al., 2001). Some of these studies evaluated passive sampler performance by collocating them with chemiluminescence analyzers during at least some portion of the field studies (Gair et al., 1991; Gair and Penkett, 1995; Plaisance et al., 2004; Kirby et al., 2001). The majority of these studies indicate that these samplers have very good precision (generally within 5%) but tend to overestimate NO_2 by 10 to 30%. However, there has not been a methodical evaluation of variables contributing to variance for the range of samplers available when used in field conditions. Thus, it is not clear if the bias is due to deviations from ideal sampling conditions that can affect actual sampling rates, contributions from co-reacting contaminants or, most probably, a combination of these variables.

A badge-type sampler was introduced by Yanagisawa and Nishimura (1982) to overcome the long sampling time required by Palmes tubes. Since then, these sensitive NO_2 short path length samplers (Toyo Roshi Ltd) have been optimized and evaluated for indoor air and for personal monitoring (Lee et al., 1993a,b). They have been used extensively for personal exposure studies (Ramirez-Aguilar et al., 2002; Yanagisawa et al., 1986; Berglund et al., 1994,

Lee et al., 2004) and indoor air measurements (Kodoma et al., 2002; Bae et al., 2004; Algar et al., 2004; Shima and Adachi, 2000; Smedje, et al., 1997) and to a more limited amount for ambient monitoring (Tashiro and Taniyama, 2002; Levy et al., 2006; Norris and Larson, 1999). Due to the greater uptake rate resulting from the larger cross sectional area of the badges and shorter diffusion length compared to the tube-type samplers, sampling times can be decreased from one-week to one-day for typical environmental air concentrations. This makes diffusive filter-badges more suitable for shorter-term sampling while long-term ambient monitoring can still be conducted using the Palmes-tubes.

Tube Type Samplers

Gradko Sampler (http://www.gradko.co.uk)

The Gradko sampler is based on the Palmes tube design (Gerboles et al., 2006b). It collects O_3 or NO_2 by molecular diffusion along an inert tube by chemisorption. A stable complex is formed with triethanolamine coated on a stainless steel screen in the tube. The complex is spectroscopically analyzed by adding an azo die (Chao and Law, 2000). The sampler has a detection limit of 0.5 ppb for NO/NO_2 and the precision of ± 6% above 5 ppb levels when used for two weeks (Table AX3.6). This sampler has been used to measure personal exposures, concentrations of residential air indoors such as in the kitchen and bedroom, and concentrations of outdoor air (Chao and Law, 2000; Gallelli et al., 2002; Lai et al., 2004). It has been used to measure ambient NO_2 levels in Southern California as a marker of traffic-related pollution in San Diego County (Ross et al., 2006).

Passam Sampler (http://www.passam.ch)

This sampler is also based on the design of the Palmes tube (Palmes et al., 1976). It collects NO_2 by molecular diffusion along an inert polypropylene tube to an absorbent, triethanolamine. The collected NO_2 is determined spectrophotometrically by the well-established Saltzmann method. When used outdoors the samplers are placed in a special shelter to protect them from rain and minimize wind turbulence effects. The Passam sampler is sold in two different models, one for long-term and one for short-term sampling.

Analyst™ Sampler (http://www.monitoreurope.com)

The Analyst™ sampler is also a modification of the open-Palmes-tube design and was developed by the Italian National Research Council (CNR – Instituto Inquinamento Atmosferico) in 2000 (Bertoni et al., 2001). The Analyst™ consists of a glass vessel, which contains a reactant supported on a stainless steel grid. It is suitable for long-term monitoring (typically one month) of oxides of nitrogen, sulfur dioxide, and volatile organic compounds in ambient air. The target compound is analyzed by gas chromatography with minimum detection limit of 0.1 mg/m^3 (~52 ppb) for a twelve-week sample duration, and has relatively high precision. The Analyst™ method development (De Santis et al., 1997, 2002) and actual field application (De Santis et al., 2004) have been described. The primary use for Analyst™ is as a reliable tool for long-term determination of concentration in indoor as well as outdoor environments (Bertoni et al., 2001) and as a screening tool for ambient monitoring to identify pollution "hot spots" (De Santis et al., 2004).

Badge-Types Samplers

Ogawa Passive Sampler (http://www.ogawausa.com)

This sampler is a double face badge that can monitor NO, NO_x, and NO_2. The design can be used also for the determination of SO_2, O_3, and NH_3 levels in air. The manufacturer-reported detection limits for nitrogen oxides are 2.3 ppb and 0.32 ppb for 24-h and 168-h sampling, respectively. Reported actual sampling rates for NO_2 are two to three times higher than the manufacturer's values. The normal operation ranges are 0 to 25 ppm for 24-h exposure and 0 to 3.6 ppm for 168-h exposure. The manufacturer recommends a sampling height of 2.5 meters and storage time of up to 1 year when kept frozen. Ogawa passive samplers have been extensively used for human exposure studies to measure personal air concentrations and (or) indoor/outdoor levels for residents in a number of locations, including adults of Richmond, Virginia (Zipprich et al., 2002), children of Santiago, Chile (Rojas-Bracho et al., 2002), office workers of Paris, France (Mosqueron et al., 2002), and cardiac compromised individuals of Toronto, Canada (Kim et al., 2006). The samplers have been used also in air monitoring networks to assess traffic-related pollutant exposure (Singer et al., 2004), as well as to evaluate spatial variability of nitrogen dioxide ambient concentrations in Montreal, Canada (Gilbert et al., 2005).

IVL Sampler (http//www.ivl.se/en/business/monitoring/diffusive_samplers.asp)

The IVL method development has been described in detail by Ferm and Svanberg (1998). It was developed by Swedish Environmental Research Institute in the mid of 1980s (Sjödin et al., 1996), is designed to minimize turbulent wind effects outdoors as well as "starvation effects" indoors (i.e., very low face velocities), interferences from within sampling tube chemistry, temperature and humidity effects, and artifacts and losses during post-sampling storage. Manufacturer-reported detection limits for this sampler with sampling times of ~1 month are 0.1 $\mu g/m^3$ (0.05 ppb) for NO_2, and 0.5 $\mu g/m^3$ (0.42 ppb) for NO, respectively. Due to its long sampling time, this sampler has been extensively used for NO_2 background monitoring in ambient air of rural or urban (Fagundez et al., 2001; Sjödin et al., 1996; Pleijel et al., 2004).

Willems Badge Sampler

The Willems badge, a short-term diffusion sampler, was developed at the University of Wageningen, Netherlands, originally for airborne ammonia measurements and later for measuring NO_2 (Hagenbjörk-Gustafsson et al., 1996). It consists of a cylinder of polystyrene with a Whatman GF-A glass fiber filter impregnated with triethanolamine at its based held in place by a 6 mm distance ring. A Teflon filter is placed on the 6 mm polystyrene ring, which is secured with a polystyrene ring of 3 mm (Hagenbjörk-Gustafsson et al., 1996). The badge is closed by a polyethylene cap to limit influences by air turbulence. The diffusion length in the badge is 6 mm. This sampler was evaluated for ambient air measurements in laboratory and field tests (Hagenbjörk-Gustafsson et al., 1999). It has a manufacturer's reported detection limit of 2 $\mu g/m^3$ (~1 ppb) for 48 h sampling duration. When used for personal sampling in an occupational setting with a minimum wind velocity of 0.3 m/s, detection limits of 18 (~9.4 ppb) and 2 $\mu g/m^3$ (~1 ppb) for 1-h and 8-h sampling, respectively, have been reported (Hagenbjörk-Gustafsson et al., 2002, Glas et al., 2004).

Radial Sampler Types

Radiello® -the radial diffusive sampler (http://www.radiello.com)

Radiello® samplers use radial diffusion over a microporous cylinder into an absorbing inner cylinder, instead of axial diffusion, which increases the uptake rate by a factor of about 100 (Hertel et al., 2001). Nitrogen dioxide is chemiadsorbed onto triethanolamine as nitrite, which is quantified by visible spectrometry. Sample collection of up to 15 days is feasible but

relative humidity higher than 70% can cause interferences when used for extended periods of more than 7 days. The manufacturer-reported typical sampling rate for nitrogen dioxide sampling is 75 ± 3.72 ml/min at temperatures between –10 and 40 °C. The rate can vary with humidity in the range of 15 to 90% and wind speed between 0.1 and 10 m/s (Radiello® Manual, 2006). A Danish study (Sørensen et al., 2005) recruited 30 subjects during each of four seasons in Copenhagen, and measured the subjects' personal exposures, home indoor/front door air concentrations during 2-day periods with this sampler.

EMD (Ecole des Mines de Douai) Sampler

A new high-uptake rate diffusive sampler has been recently developed by the Ecole des Mines de Douai (EMD) laboratory (Piechocki-Minguy et al., 2003) and evaluated in the laboratory and field for measurement of NO_2 levels in ambient air. It is composed of a porous cartridge impregnated with triethanolamine and fitted in a cylindrical protective box equipped with caps at its extremities (Piechocki-Minguy et al., 2006). The large sampling area (cartridge surface) and the two circular openings provide a high uptake rate (exceeding 50 cm^3/min). The sampling rate was reported to be on average 0.89 cm^3/s for indoor sampling and 1.00 cm^3/s for outdoor sampling. Detection limits were determined to be 11 $\mu g/m^3$ (~5.8 ppb) for 1-h measurement. The sampling rate was not significantly influenced by wind at speeds higher than 0.3 m/s (Piechocki-Minguy et al., 2003). This sampler has been used in France to assess personal exposures in a series of microenvironments (home, other indoor places, transport and outdoor) for two 24-h time periods (weekday and weekend) (Piechocki-Minguy et al., 2006).

NO_2 Measurements in Epidemological Studies

Since passive samplers are the most frequently used monitoring method in epidemiology studies of NO_2 effects, their performance compared to the long established chemiluminescence monitoring method is critical for determining the contribution of measurement error to exposure estimates. First, most passive samplers developed and used for personal and indoor exposure studies need to be employed for at least 24 h to collect sufficient NO_2 to be detected. Therefore, the majority of measurements of personal exposure concentrations done to date represents daily or longer integrated or average exposure and cannot be used to assess acute, peak exposure concentrations. Some newer passive samplers for nitrogen dioxide have higher uptake rates and active pump samplers with traditional battery operated sampling pumps and appropriate

adsorbents can collect sufficient NO_2 in approximately one h and have been used in a few studies providing information on exposure in microenvironments and shorter term exposure concentration. Hourly fluctuations in nitrogen dioxide concentrations may be important to the evaluation of exposure-health effects relationship, so continuous monitors, such as those used at central site monitoring stations are still the only approach for estimating short-term exposures.

Second, interferences for other nitrogen oxide species can contribute to NO_2 exposure monitoring errors. Both the chemiluminescence analyzer and passive samplers experience these interferences but the kinetics and stoichiometry of interferent compound reactions have not been well established, especially for the passive samplers. As indicated earlier, TEA-based diffusive sampling methods tend to overestimate NO_2 concentrations in field comparisons with chemiluminescence analyzers. This could be in part the result of chemical reactions between ozone and nitric oxide (NO) within the diffusion tube, leading to as much as an overestimate up to 30%, or differential sensitivity to other nitrogen oxides between the passive and active samplers. Due to spatially and temporally variability of NO and NO_2 concentrations, especially at roadsides where nitric oxide concentrations are relatively high and when sufficient ozone is present for interconversion between the species, lack of agreement between the passive sampler and central continuous monitor can represent differences in sampler response (Heal et al., 1999, Cox, 2003). In the U.K., an alternative nitrogen dioxide monitoring plan using cost-effective and simpler tube-type passive sampler has been proposed and implemented countrywide. However, careful investigation of nitrogen dioxide levels revealed an overestimation, around 30% by the passive sampler (Campbell et al., 1994). Another evaluation study (Bush et al., 2001) showed that the overall average NO_2 concentrations calculated from diffusion tube measurements were likely to be within 10% of chemiluminescent measurement data.

Third, the effect of environmental conditions (e.g., temperature, wind speed, and humidity) on the performance of passive samplers is still a concern when using it for residential indoor, outdoor, and personal exposure studies, because of sampling rates that deviate from ideal and can vary through the sampling period. Overall, field test results of passive sampler performance are not consistent and they have not been extensively studied over a wide range of concentrations, wind velocities, temperatures and relative humidities (Varshney and Singh, 2003). Therefore, studies directed at investigating the contributions from environmental conditions to the performance of diffusive samplers in multiple locations need to be undertaken.

AX3.4 NITROGEN OXIDES IN INDOOR AIR

AX3.4.1 Indoor Sources and Concentrations of Nitrogen Oxides

Penetration of outdoor NO_2 and combustion in various forms are the major sources of NO_2 to indoor environments. These environments include homes, schools, restaurants, theaters etc. As might be expected, indoor concentrations of NO_2 in the absence of combustion sources are determined by the infiltration of outdoor NO_2 (Spengler et al., 1994; Weschler et al., 1994; Levy et al., 1998a), with a much smaller contribution from chemical reactions in indoor air. Indoor sources of nitrogen oxides have been characterized in several reviews, namely the last AQCD for Oxides of Nitrogen (U.S. Environmental Protection Agency, 1993); the Review of the Health Risks Associated with Nitrogen Dioxide and Sulfur Dioxide in Indoor Air for Health Canada (Brauer et al., 2002); and the Staff Recommendations for revision of the NO_2 Standard in California (CARB, 2006). Mechanisms by which nitrogen oxides are produced in the combustion zones of indoor sources were reviewed in the last AQCD for Oxides of Nitrogen (U.S. Environmental Protection Agency, 1993) and will not be repeated here. Sources of ambient NO_2 are reviewed in Chapter 2 of this document. It should also be noted that indoor sources can affect ambient NO_2 levels, particularly in areas in which atmospheric mixing is limited.

Because most people spend most of their time indoors, personal exposure is primarily determined by indoor air quality as shown in Figure AX3.21. Ideally, exposure to NO_2 should be cumulated over all indoor environments in which an individual spends time. These indoor environments may include homes, schools, offices, restaurants, theaters, ice skating rinks, stores, etc. However, in a study by Leaderer et al. that used two-week integrated measures, concentrations of NO_2 inside the home accounted for 80% of the variance in total personal exposure, indicating that home concentrations are a reasonable proxy for personal exposure (Leaderer et al., 1986).

Homes

Combustion of fossil and biomass fuels produce nitrogen oxides and the importance of such sources for determining human exposures depends on how emissions are allowed to mix into living areas and whether emissions are vented to the outdoors or not. Combustion of fossil

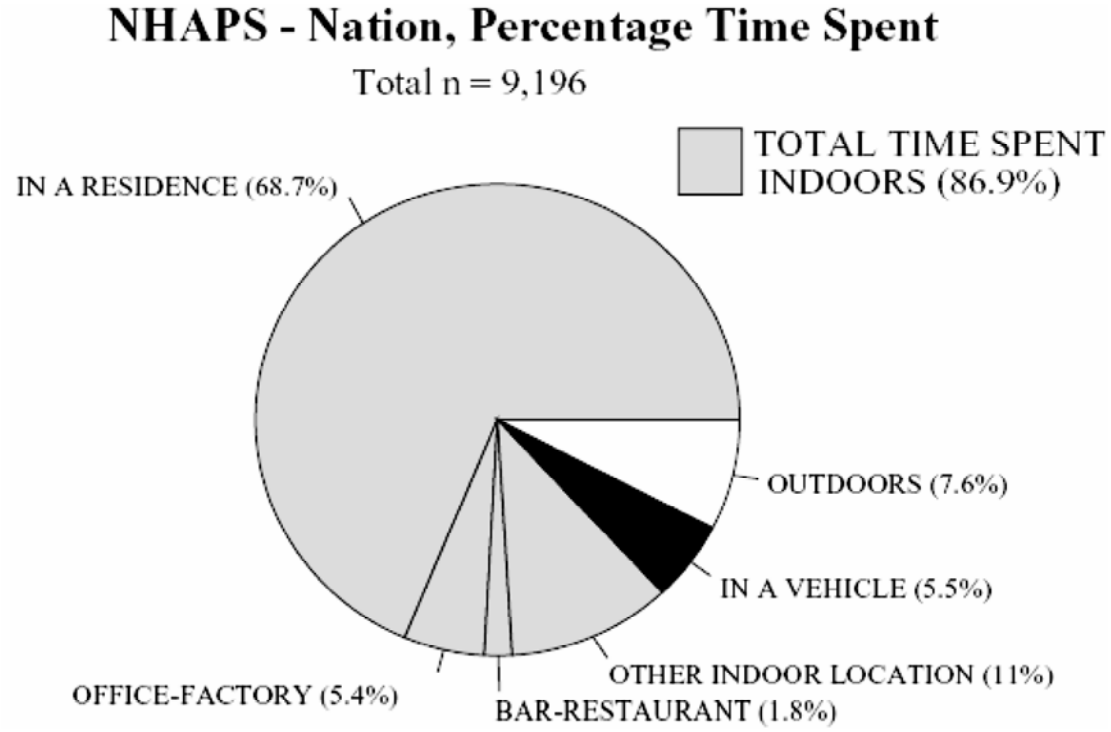

Figure AX3.21. Percentage of time people spend in different environments.

Source: Klepeis et al. (2001).

fuels occurs in gas-fired appliances used for cooking, heating, and drying clothes; oil furnaces; kerosene space heaters; and coal stoves. Motor vehicles and various types of generators also contribute in structures attached to living areas. Biomass fuels include mainly wood used in fireplaces and wood stoves and tobacco.

Gas Cooking Appliances

A large number of studies, as described in the reviews cited above, have all noted the importance of gas cooking appliances as sources of NO_2 emissions. Depending on geographical location, season, other sources, length of monitoring period, and household characteristics, homes with gas cooking appliances have approximately 50% to over 400% higher NO_2 concentrations than homes with electric cooking appliances (Gilbert et al., 2006; Lee et al., 2002; Lee et al., 2000; García-Algar et al., 2004; Raw et al., 2004; Leaderer et al., 1986; García-Algar, 2003). Gas cooking appliances remain significantly associated with indoor NO_2 concentrations

after adjusting for several potential confounders including season, type of community, socioeconomic status, use of extractor fans, household smoking, and type of heating (García-Algar et al., 2004; Garrett, 1999).

Gas appliances with pilot lights emit more NO_2 than gas appliances with electronic ignition. Spengler et al. (1994) found that NO_2 concentrations in bedrooms of homes with a gas range without a pilot light averaged 4 ppb higher than in homes with an electric range, but were 15 ppb higher in homes with gas ranges with pilot lights. Lee et al. (1998) found somewhat larger differences in NO_2 concentrations in homes in the Boston area, with minor seasonal variation. Homes with gas stoves without pilot lights averaged between 11 ppb (summer) and 18 ppb (fall) higher than homes with electric stoves, while those with pilot lights averaged between 19 ppb (summer) and 27 ppb (fall) higher than electric stove homes.

Use of extractor fans reduces NO_2 concentrations in homes with gas cooking appliances (Gallelli et al., 2002; García-Algar et al., 2003), although absolute NO_2 levels tend to remain higher than in homes with electric stoves. In a multivariate analysis, García-Algar et al. (2004) found that having a gas cooker remained significantly increased NO_2 concentrations even after adjusting for extractor fan use. Raw et al. (2004) found only a small effect of extraction fan use on NO_2 levels in the bedroom in gas cooker homes. Among homes with gas cooking, geometric mean bedroom NO_2 levels were 1.7 ppb lower in homes with an extractor fan than in homes without one. As expected, among homes with no fossil fuel cooking, there were no differences in mean bedroom levels of NO_2 in homes with and without extractor fans.

Other Combustion Sources

Secondary heating appliances are additional sources of NO_2 in indoor environments, particularly if they are unvented or inadequately vented. As heating costs increase, the use of these secondary heating appliances tends to increase. From 1988 to 1994, an estimated 13.7 million homes used unvented heating appliances, with disproportionately higher usage rates among southern, rural, low-income, and African-American homes (Slack and Heumann, 1997). Of the 83.1 million households using gas stoves or ovens for cooking, 7.7 million (9.3%) also used the stove for heating (Slack and Heumann, 1997).

Gas heaters, particularly when unvented or inadequately vented, produce high levels of NO_2. Kodoma et al. (2002) examined the associations between secondary heating sources and NO_2 concentrations measured over a 48-h exposure period in the living rooms of homes in

Tokyo, Japan. They found much higher NO_2 concentrations during February 1998 and January 1999 in homes with kerosene heaters in both southern (152.6 ppb and 139.7 ppb for 1998 and 1999, respectively) and northern (102.4 and 93.1 ppb for 1998 and 1999, respectively) areas of Tokyo compared to homes with electric heaters (30.8 and 31.1 for the southern and 37.2 and 31.6 for northern areas, 1998 and 1999, respectively).

In a study by Garrett et al. (1999) of 78 homes in Latrobe Valley, Australia, the two highest indoor NO_2 levels recorded in the study were 129 ppb for the only home with an unvented gas heater and 69 ppb for a home with a vented gas heater. Levels of NO_2 in the kitchens and living rooms of homes with a vented gas heater (mean = 6.9 ppb in living room, 7.3 ppb in kitchen, n = 15) were comparable to homes with gas stoves (mean = 6.7 ppb in living room, 8.0 ppb in kitchen, n = 15) (Table AX3.7). These concentrations include results from all seasons combined, so the levels are somewhat lower than those found by Triche et al. (2005) for winter monitoring periods only.

Triche et al. (2005) also found high levels of NO_2 in homes with gas space heaters, although information on whether the appliance was vented or unvented was not available. Data from this study were analyzed in more detail and are shown in Table AX3.8. The median NO_2 concentration in the 6 homes with gas space heater use during monitoring periods with no gas stove use was 15.3 ppb; a similar incremental increase in total NO_2 levels was noted for homes with gas space heater use during periods when gas stoves were also used (Median = 36.6 ppb) compared to homes where gas stoves were used but no secondary heating sources were present (Median = 22.7 ppb) (Table AX3.8).

Shima and Adachi (1998) examined associations between household characteristics, outdoor NO_2, and indoor NO_2 in 950 homes during the heating season (640 with unvented and 310 vented heaters) and 905 homes during the non-heating season in urban, suburban, and rural areas of Japan. While no information is provided on gas stove use, the authors note that nearly all homes in Japan have gas stoves, though relatively few have pilot lights. During the heating season, geometric mean NO_2 levels in homes with unvented heaters (66.4 ppb) are about three times higher than in homes with vented heaters (20.6 ppb). In the non-heating season, the mean levels were lower at only 13.8 ppb, suggesting a contribution from vented heaters as well.

In multivariate analyses, Gilbert et al. (2006) found that gas and mixed/other heating systems were significantly associated with NO_2 levels, adjusting for presence of gas stoves and air exchange rates in 96 homes in Quebec City, Canada during the winter/early spring period. Many homes with gas space heaters also have gas stoves, and the contribution from multiple sources is much higher than from any single source alone (Garrett et al., 1999). In the Garrett et al. (1999) study, homes were classified into five categories: no indoor source (n = 15), gas stove only (n = 15), gas heater only (n = 14), smoker in the household only (n = 7), and multiple sources (n = 29). Homes with multiple sources had much higher NO_2 concentrations homes with either a gas stove only or gas heater only (Table AX3.9).

Kerosene heaters are also important contributors to indoor NO_2 levels. Leaderer et al. (1986) enrolled a cohort of kerosene heater users identified from local kerosene dealers and a cohort of controls systematically chosen from the same neighborhoods with each matched pair treated as a sampling unit (i.e., sampled at the same randomly assigned time period). A total of 302 homes were monitored for at least one two-week period. While outdoor concentrations never exceeded 100 $\mu g/m^3$ (53 ppb), approximately 5% of homes with either no gas but 1 kerosene heater or gas but no kerosene heater had levels exceeding 53 ppb. Between 17%-33% of homes with both gas and kerosene heater(s) exceeded this limit, while nearly one quarter of homes with no gas, but two or more kerosene heaters had these levels.

Data from Triche et al. (2005) (Table AX3.8) also indicated increased levels of NO_2 for kerosene heater homes during monitoring periods with no gas stove use (Median = 18.9 ppb) compared to homes with no sources (Median = 6.3 ppb), which is similar to levels found in homes using gas space heaters (Median = 15.3 ppb). However, these NO_2 concentrations are of the same magnitude as those in homes with gas stove use (Median = 17.2 ppb).

Data are available for unvented gas hot water heaters from a number of studies conducted in the Netherlands. Results summarized by Brauer et al. (2002) indicate that concentrations of NO_2 in homes with unvented gas hot water heaters were 10 to 21 ppb higher than in homes with vented heaters, which in turn, had NO_2 concentrations 7.5 to 38 ppb higher than homes without gas hot water heaters.

The contribution from combustion of biomass fuels has not been studied as extensively as that from gas. A main conclusion from the previous AQCD was that properly vented wood stoves and fireplaces would make only minor contributions to indoor NO_2 levels. Several studies

conclude that use of wood burning appliances does not increase indoor NO_2 concentrations. Levesque et al. (2001) examined the effects of wood-burning appliances on indoor NO_2 concentrations in 49 homes in Quebec City, Canada. The homes, which had no other combustion source, were sampled for 24 h while the wood-burning appliance was being used. No significant differences in mean NO_2 levels were found in homes with (6.6 + 3.6 ppb) and without (8.8 + 1.9 ppb) a wood-burning appliance. Data from Triche et al. (2005) confirm these findings (Table AX3.8). Homes with wood burning sources had comparable NO_2 concentrations to homes without other secondary heating sources, with (Median = 5.9 ppb) and without (Median = 16.7 ppb) gas stove use.

Table AX3.9 shows short-term average (minutes to a few hours) concentrations of NO_2 in homes with combustion sources. The concentrations represent those found in different rooms in houses sampled. However, concentrations are much higher in those persons directly exposed to emissions. For example, Dennekamp et al. (2001) found NO_2 concentrations of about 1 ppm at face level in front of a 4-burner gas range. Table AX3.10 shows long-term average (24-h to 2 week) concentrations of NO_2 in homes with combustion sources (mainly gas fired).

Data are available for unvented gas hot water heaters from a number of studies conducted in the Netherlands. Results summarized by Brauer et al. (2002) indicate that concentrations of NO_2 in homes with unvented gas hot water heaters were 10 to 21 ppb higher than in homes with vented heaters, which in turn, had NO_2 concentrations 7.5 to 38 ppb higher than homes without gas hot water heaters.

As can be seen from the tables, shorter-term average concentrations tend to be much higher than longer term averages. However, as Triche et al. (2005) point out, the 90th percentile concentrations can be substantially greater than the medians, even for two week long samples.

This finding illustrates the high variability found among homes. This variability reflects differences in ventilation of emissions from sources, air exchange rates, the size of rooms etc. The concentrations for short averaging periods that are listed in Table AX3.9 correspond to about 10 to 30 ppb on a 24-h average basis. As can be seen from inspection of Table AX3.10, these sources would contribute significantly to the longer term averages reported there if operated on a similar schedule on a daily basis. This implies that measurements made with long averaging periods may not capture the nature of the diurnal pattern of indoor concentrations in

homes with strong indoor sources. This problem becomes more evident as ambient NO_2 levels decrease due to more efficient controls on outdoor sources.

In 10% of homes with fireplaces studied by Triche et al. (2005), NO_2 concentrations were greater than or equal to 80 ppb, or about twice the level found in homes with no indoor combustion source (see Figure AX3.30). In a study of students living in Copenhagen, Sørensen et al. (2005) found that personal exposures to NO_2 were significantly associated with time exposed to burning candles in addition to other sources. However, they did not provide data for concentrations in spaces in which candles were burned. Results of studies relating NO_2 concentrations and exposures to environmental tobacco smoke (ETS) have been mixed. Several studies found positive associations between NO_2 levels and ETS (e.g., Linaker et al., 1996); Farrow et al., 1997; Alm et al., 1998; Levy et al., 1998a; Monn et al., 1998; Cyrys et al., 2000; Lee et al., 2000; García-Algar, 2004) whereas others have not (e.g., Hackney et al., 1992; Kawamoto et al., 1993). In a study of 57 homes in Brisbane, Australia (Lee et al., 2000), levels of NO_2 were higher in homes with smokers present (14.9 + 7.7 ppb) than without smokers (9.9 + 5.0 ppb). However, these concentrations did not account for presence of a gas range (n = 18 of 57 homes had a gas range). Garrett et al. (1999) found that smoking in the home increased levels of NO_2 in the winter, but not in the summer when windows tended to be opened. In a study of students living in Copenhagen, Sørensen et al. (2005) did not find a significant association between ETS and personal exposures to NO_2. However, they found that burning candles was a significant prediction of bedroom levels of NO_2.

Other Indoor Environments

Indoor ice skating rinks have been cited as environments containing high levels of NO_2 when fuel powered ice resurfacing machines are used especially without ventilation. As part of a three year study, Levy et al. (1998b) measured NO_2 concentrations at 2 locations at the outside of the ice surface in 19 skating rinks in the Boston area over 3 winters. Although different passive samplers were used in the first year (Palmes tubes, 7 day sampling time) and in years 2 and 3 (Yanagisawa badges, 1 day working hours) of the study, consistently high mean NO_2 concentrations were associated with the use of propane fueled resurfacers (248 ppb in the first year and 206 ppb in the following years) and gasoline fueled resurfacers (54 ppb in the first year and 132 ppb in the following years) than with electric resurfacers (30 ppb in the first year and 37 ppb in the following years). During all three years of the study peak NO_2 concentrations were

several times higher in the rinks with propane and gasoline fueled resurfacers than the values given above. A number of earlier studies have also indicated NO_2 concentrations of this order and even higher (Paulozzi et al., 1993; Berglund et al., 1994; Lee et al., 1994; Brauer et al., 1997). In these studies peak averages were in the range of a few ppm.

AX3.4.2 Reactions of NO_2 in Indoor Air

Chemistry in indoor settings can be both a source and a sink for NO_2 (Weschler and Shields, 1997). NO_2 is produced by reactions of NO with ozone or peroxy radicals, while NO_2 is removed by gas phase reactions with ozone and assorted free radicals and by surface promoted hydrolysis and reduction reactions. The concentration of indoor NO_2 also affects the decomposition of peroxyacyl nitrates. Each of these processes is discussed in the following paragraphs. They are important not only because they influence the indoor NO_2 concentrations to which humans are exposed, but also because certain products of indoor chemistry may confound attempts to examine associations between NO_2 and health.

Indoor NO can be oxidized to NO_2 by reaction with ozone or peroxy radicals; the latter are generated by indoor air chemistry involving O_3 and unsaturated hydrocarbons such as terpenes found in air fresheners and other household products (Sawar et al., 2002a,b; Nazaroff and Weschler, 2004; Carslaw, 2007). The rate coefficient for the reaction

$$O_3 + NO \rightarrow NO_2 + O_2 \qquad \text{(AX3-5)}$$

at room temperature (298 K) is 1.9×10^{-14} cm^3/molec-sec or 4.67×10^{-4} ppb^{-1} s^{-1} (Jet Propulsion Laboratory, 2006). At an indoor O_3 concentration of 10 ppb and an indoor NO concentration that is significantly less than that of O_3, the half-life of NO is 2.5 min. This reaction is sufficiently fast to compete with even relatively fast air exchange rates. Hence, the amount of NO_2 produced from NO tends to be limited by the amount of O_3 available. The indoor concentrations of NO and O_3 are negatively correlated; significant concentrations of NO can only accumulate when small amounts of O_3 are present and vice versa (Weschler et al., 1994).

The rapid reaction between NO and O_3 also means that humans, themselves, can be indirect sources of NO_2 in the rooms they occupy. Exhaled human breath contains NO that is generated endogenously (Gustafsson et al., 1991). For a typical adult male, the average nasal

NO output is 325 nL min^{-1} or 23.9 μg h^{-1} (Imada et al., 1996). If ozone is present in the indoor air, some or all of these exhaled NO molecules will be oxidized to NO_2. To put this source in perspective, consider the example of an adult male in a 30 m^3 room ventilated at 1 air change per hour (h-1) with outdoor air. The steady-state concentration of NO in the room as a consequence of NO in exhaled breath is 0.80 μg m^3 or 0.65 ppb if none of the NO were to be oxidized. However, assuming a meaningful concentration of ozone in the ventilation air (>5 ppb), most of this NO is oxidized to NO_2 before it is exhausted from the room. In this scenario, the single human occupant is indirectly a source for 0.65 ppb of NO_2 in the surrounding air. At higher occupant densities, lower air exchange rates and elevated concentrations of O_3 in the ventilation air, human exhaled breath could contribute as much as 5 ppb to the total concentration of indoor NO_2.

The reaction of NO_2 with ozone produces nitrate radicals (NO_3):

$$O_3 + NO_2 \rightarrow NO_3 + O_2 \qquad \text{(AX3-6)}$$

The second order rate-constant for this reaction at room temperature (298 K) is 3.2×10^{-17} cm^3/molec-sec or 7.9 x 10^{-7} ppb^{-1} s^{-1} (Jet Propulsion Labatory, 2006). For indoor concentrations of 20 ppb and 30 ppb for O_3 and NO_2, respectively, the production rate of NO_3 radicals is 1.7 ppb h^{-1}. This reaction is strongly temperature dependent, an important consideration given the variability of indoor temperatures with time of day and season. The nitrate radical is photolytically unstable (Finlayson-Pitts and Pitts, 2000). As a consequence, it rapidly decomposes outdoors during daylight hours. Indoors, absent direct sunlight, nitrate radical concentrations may approach those measured during nighttime hours outdoors. To date there have been no indoor measurements of the concentration of nitrate radicals in indoor settings. Modeling studies by Nazaroff and Cass (1986), Weschler et al. (1992), Sarwar et al. (2002b), and Carslaw et al. (2007) estimate indoor nitrate radical concentrations in the range of 0.01 to 5 ppt, depending on the indoor levels of O_3 and NO_2.

The nitrate radical and NO_2 are in equilibrium with dinitrogen pentoxide (N_2O_5):

$$NO_3 + NO_2 \leftrightarrow N_2O_5 \qquad \text{(AX3-7)}$$

Dinitrogen pentoxide reacts with water to form nitric acid. The gas phase reaction with water is too slow (Sverdrup et al., 1987) to compete with air exchange rates in most indoor environments.

Due to mass transport limits on the rate at which N_2O_5 is transported to indoor surfaces, reactions of N_2O_5 with water sorbed to indoor surfaces are much slower than gas phase reactions between nitrate radicals and commonly occurring indoor alkenes.

Once formed, NO_3 radicals can oxidize organic compounds by either adding to an unsaturated carbon bond or abstracting a hydrogen atom (Wayne et al., 1991). In certain indoor settings, the nitrate radical may be a more important indoor oxidant than either ozone or the hydroxyl radical. Table 8 in Nazaroff and Weschler (2004) illustrates this point. Assuming indoor concentrations of 20 ppb, 5×10^{-6} ppb, and 0.001 ppb for O_3, OH, and NO_3, respectively, the pseudo first-order rate constants for reactions of most terpenoids are larger for reactions with NO_3 than for reactions with either O_3 or OH. For example, for the stated conditions, the half-lives of d-limonene and α-pinene are roughly three times shorter as a consequence of reaction with NO_3 versus reaction with O_3. The products of reactions between NO_3 and various organic compounds include nitric acid, aldehydes, ketones, organic acids and organic nitrates; these have been summarized by Wayne et al. (1991). Nitrate radicals and the products of nitrate radical chemistry may be meaningful confounders in NO_2 exposure studies.

Reactions between NO_2 and various free radicals can be an indoor source of organo-nitrates, analogous to the chain-terminating reactions observed in photochemical smog (Weschler and Shields, 1997). Additionally, based on laboratory measurements and measurements in outdoor air (Finlayson-Pitts and Pitts, 2000), one would anticipate that NO_2, in the presence of trace amounts of HNO_3, can react with PAHs sorbed on indoor surfaces to produce mono- and dinitro-PAHs.

As noted earlier in Chapter 2, HONO occurs in the atmosphere mainly via multiphase processes involving NO_2. HONO is observed to form on surfaces containing partially oxidized aromatic structures (Stemmler et al., 2006) and on soot (Ammann et al., 1998). Indoors, surface-to-volume ratios are much larger than outdoors, and the surface mediated hydrolysis of NO_2 is a major indoor source of HONO (Brauer et al., 1990; Febo and Perrino, 1991; Spicer et al., 1993; Brauer et al., 1993; Spengler et al., 1993; Wainman et al., 2001; Lee et al., 2002). Spicer et al. (1993) made measurements in a test house that demonstrated HONO formation as a consequence of NO_2 surface reactions and postulated the following mechanism to explain their observations:

$$2NO_2 + H_2O/surface \rightarrow HONO(aq) + H^+ + NO_3^- \quad \text{(AX3-8)}$$

$$HONO(aq) \leftrightarrow HONO(g) \quad \text{(AX3-9)}$$

In a series of chamber studies, Brauer et al. (1993) reported HONO formation as a consequence of NO_2 surface reactions and further reported that HONO production increased with increasing relative humidity. Wainman et al. (2001) confirmed Brauer's findings regarding the influence of relative humidity. They also found that NO_2 removal and concomitant HONO production was greater on synthetic carpet surfaces compared to Teflon surfaces, and that the affinity of a surface for water influences HONO's desorption from that surface. Lee et al. (2002) measured HONO and NO_2 concentrations in 119 Southern California homes. Average indoor HONO levels were about 6 times larger than outdoors (4.6 ppb versus 0.8 ppb). Indoor HONO concentrations averaged 17% of indoor NO_2 concentrations, and the two were strongly correlated. Indoor HONO levels were higher in homes with humidifiers compared to homes without humidifiers (5.9 ppb versus 2.6 ppb). This last observation is consistent with the studies of Brauer et al. (1993) and Wainman et al. (2001) indicating that the production rate of HONO from NO_2/surface reactions is larger at higher relative humidities. Based on detailed laboratory studies, the hydrolysis mechanism, Equations AX3-8 and AX3-9, have been refined. Finlayson-Pitts et al. (2003) hypothesize that the symmetric form of the NO_2 dimer is sorbed on surfaces, isomerizes to the asymmetric dimer which auto ionizes to $NO^+NO_3^-$; the latter then reacts with water to form HONO and surface adsorbed HNO_3. FTIR-based analyses indicate that the surface adsorbed HNO_3 exists as both undissociated nitric acid-water complexes, $(HNO_3)_x(H_2O)_y$, and nitrate ion-water complexes, $(NO_3^-)_x(H_2O)_y$ (Dubowski et al., 2004, Ramazan et al., 2006). Such adsorbed species may serve as oxidizing agents for organic compounds sorbed to these same surfaces (Ramazan et al., 2006).

HONO and much smaller amounts of HNO_3 are also emitted directly by combustion by gas appliances and can infiltrate from outdoors. Spicer et al. (1993) compared the measured increase in HONO in a test house resulting from direct emissions of HONO from a gas range and from production by surface reactions of NO_2. They found that emissions from the gas range could account for about 84% of the measured increase in HONO and surface reactions for 11% in an experiment that lasted several hours. An equilibrium between adsorption of HONO from the gas range (or other indoor combustion sources) and HONO produced by surface reactions

(see Equation AX3-9) also determines the relative importance of these processes in producing HONO in indoor air. In a study of Southern CA homes (Lee et al., 2002), indoor levels of NO_2 and HONO were positively associated with the presence of gas ranges.

It is known that the photolysis of HONO (g) in the atmosphere (outdoors) is a major source of the hydroxyl radical (OH). Given high indoor HONO concentrations and the presence of lighting (sun light penetrating windows, incandescent lights, fluorescent lights), the photolysis of indoor HONO may be a meaningful source of indoor hydroxyl radical, under favorable reaction conditions. Given the large suite of man-made chemicals present indoors at elevated concentrations, indoor free radicals (e.g., OH and NO_3) can initiate and drive a complex series of indoor chemical reactions.

NO_2 can also be reduced on certain surfaces, forming NO. Spicer et al. (1989) found that as much as 15% of the NO_2 removed on the surfaces of masonite, ceiling tile, plywood, plasterboard, bricks, polyester carpet, wool carpet, acrylic carpet and oak paneling was re-emitted as NO. Weschler and Shields (1996) found that the amount of NO_2 removed by charcoal building filters were almost equally matched by the amount of NO subsequently emitted by these same filters.

Spicer et al. (1993) determined the 1st order rate constants for removal of several NO_y components by reaction with indoor surfaces. They found lifetimes (e-folding times) of about half an hour for HNO_3, an hour for NO_2, and hours for NO and HONO. Thus the latter two components, if generated indoors are more likely to be lost to the indoor environment through exchange with outside air than by removal on indoor surfaces. However, HONO is in equilibrium with the nitrite ion (NO_2^-) in aqueous surface films:

$$HONO(aq) \leftrightarrow H^+ + NO_2^- \qquad (3\text{-}10)$$

Ozone oxidation of nitrite ions in such films is a potential sink for indoor HONO (Lee et al., 2002).

Jakobi and Fabian (1997) measured indoor and outdoor concentrations of ozone and peroxyacetyl nitrate (PAN) in several offices, private residences, a classroom, a gymnasium and a car. They found that indoor levels of PAN were 70% to 90% outdoor levels, and that PAN's indoor half-life ranged from 0.5 to 1 h. The primary indoor removal process is thermal decomposition:

$$CH_3C(O)OONO_2 \leftrightarrow CH_3C(O)OO + NO_2 \quad \text{(AX3-11)}$$

As is indicated by Equation AX3-11, PAN is in equilibrium with the peroxylacetyl radical and NO_2. Hence, the indoor concentration of NO_2 affects the thermal decomposition of PAN and, analogously, other peroxyacyl nitrates. Peroxylalkyl radicals rapidly oxidize NO to NO_2, so the indoor concentration of NO also influences the thermal decomposition of PAN type species (Finlayson-Pitts and Pitts, 2000).

Reactions between hydroxyl radicals and aldehydes in the presence of NO_2 can lead to the formation of peroxyacyl nitrates. Weschler and Shields (1997) have speculated that such chemistry may sometimes occur indoors. For example, the requisite conditions for the formation of the highly irritating compound peroxybenzoyl nitrate may occur when ozone, certain terpenes, styrene and NO_2 are present simultaneously at low air exchange rates. This relatively common indoor mixture of pollutants produces hydroxyl radicals and benzaldehyde, which can subsequently react as noted above. In her detailed model of indoor chemistry, Carslaw (2007) explores the indoor formation of PAN-type species (see Figure 2 in the cited reference).

Recent work indicates that indoor NO_2 also can affect the formation of secondary organic aerosols (SOA) resulting from the reaction of O_3 with terpenes such as d-limonene and α-pinene (Nøjgaard et al., 2006). At concentrations of 50 ppb for O_3 and the terpenes, NO_2 decreased the formation of SOA compared to the levels formed in the absence of NO_2. The effect was more pronounced for SOA derived from α-pinene than d-limonene, and at lower NO_2 concentrations, appears to be explained by the O_3 loss resulting from its reaction with NO_2. The resultant nitrate radicals apparently are not as efficient at producing SOA as the lost O_3.

Nitro-PAHs have been found in indoor environments (Mumford et al., 1991; Wilson et al., 1991). The major indoor sources of nitro-PAHs include cooking, wood burning, and the use of kerosene heater (World Health Organization (WHO), 2003). It is also likely that nitro-PAHs outdoors can infiltrate indoors. One of the potential sources of nitro-PAHs indoors, which has not been characterized, is reactions via indoor chemistry. The reactions of PAHs with OH and NO_3 may occur in indoor environments. Although no direct measurements of OH or NO_3 in indoor environments, OH and NO_3 can be formed via indoor chemistry and may present at significant levels indoors (Nazaroff and Cass 1986, Sarwar et al., 2002a; Carslaw, 2007). Concentrations of ~10^{-6} ppb for OH and 0.01-5 ppt of NO_3 have been predicted through indoor

chemical reactions (Nazaroff and Cass 1986, Sarwar et al., 2002a, Carslaw, 2007), depending on the indoor levels of O_3, alkenes, and NO_2. Observation of secondary organic aerosols (SOA) formation in a simulated indoor environment also suggested that ~10^{-5} ppb steady-state OH radicals were generated from the reactions of O_3 with terpenes (Fan et al., 2003). PAHs are common indoor air pollutants (Chuang et al., 1991; Naumova et al., 2002), and the concentrations of some PAHs indoors are often higher than outdoors (Naumova et al., 2002). Therefore, the reactions of OH and NO_3 with PAHs may occur at rates comparable to air exchange rates to form nitro-PAHs indoors. In addition, the reactions of NO_3 with PAHs may be more significant indoors than outdoors because indoor NO_3 is more stable due to the low uv in indoor environments. Given the high surface areas available indoors, the formation of nitro-PAHs via surface reactions of PAHs with nitrating species may be more important compared to heterogeneous reactions outdoors.

In summary, indoor chemistry can meaningfully alter the indoor concentration of NO_2. Indoor exposure to NO_2 may be accompanied by indoor exposures to nitrate radicals, organic nitrates, and nitro-PAHs.

AX3.4.3 Contributions from Outdoor NO_2

As might be expected, indoor concentrations of NO_2 in the absence of combustion sources are primarily determined by outdoor NO_2 concentrations (Spengler et al., 1994; Weschler et al., 1994; Levy et al., 1998a), with a much smaller contribution from chemical reactions in indoor air.

The exchange between NO_2 in ambient air and in the indoor environment is influenced by infiltration (air leakage), natural ventilation (air flow through intentional openings such as windows), and mechanical ventilation (rarely used in residences) (Yang et al., 2004). In temperate climates, winter is associated with lower indoor/outdoor ratios of NO_2 since windows and doors are usually tightly closed and the only source of exchange is infiltration. Newer homes tend to be built more tightly than older homes, so have even lower rates of infiltration. During warmer weather, air conditioner use and opening of windows increase air exchange between outdoors and indoors.

Yang et al. (2004) used multiple integrated (7-day) NO_2 measurements indoors and outdoors to calculate penetration and source strength factors in Seoul, Korea and Brisbane,

Australia using a mass balance model considering a residence as a single chamber (Yang et al., 2004). They showed that, while penetration factors did not differ significantly between gas and electric range homes, source strength factors were much higher in homes with gas ranges in both Brisbane and Seoul (5.77 ± 3.55 and 9.12 ± 4.50, respectively) than in electric range homes in Brisbane (1.49 ± 1.25). Similarly, calculated NO_2 source strengths ($\mu g/m^3/h$) were 21.9 ± 21.8 and 44.7 ± 38.1 in gas homes in Brisbane and Seoul, respectively, and 6.6 ± 6.3 in electric homes in Brisbane.

Household Characteristics

Yang et al. (2004) found that levels of indoor NO_2 (in $\mu g/m^3$) were associated with house characteristics in 28 homes in Brisbane (where there were both electric and gas range homes). Homes with a gas water heater had higher levels than those without (34.5 ± 16.4 versus 22.8 ± 12.1, p = 0.048), but these were unadjusted associations, and it is likely that many of the homes with gas water heaters also had gas ranges. Homes with an attached garage had higher levels of NO_2 (33.1 ± 18.3) compared to homes without one (21.8 ± 8.8) (p = 0.039). Attached garages were not, however, associated with NO_2 levels in a study in Quebec City, Canada (Gilbert et al., 2006). The authors suggested that the lack of association might be attributed to small numbers (n = 18 homes with attached garages) or to the airtightness of homes in Canada compared to those in Australia.

Location in a city center was associated with higher NO_2 levels in homes in Menorca (one of the Balearic Islands off the coast of Spain with rural and small town residences), after adjusting for gas cooker, extractor fan use, smoking in the home, type of central heating, season, and social class (García-Algar., 2004). In the same study, levels of indoor NO_2 in Barcelona (a large coastal city in Spain) and Ashford (a medium-sized town in the southeast UK) were significantly higher than those in Menorca

In a study of a random sample of 845 homes in England (Raw et al., 2004), levels of NO_2 were significantly associated with dwelling type and age of home, but the authors attributed these effects to the geographical location of the home (e.g., inner city). Garrett et al. (1999) also found that age of house was significantly associated with NO_2 levels in winter and summer. In the study by Shima and Adachi, (1998), differences in concentrations of NO_2 between homes with and without unvented heaters in the heating season were slightly lower among homes with wood compared to aluminum window frames. Type of window frames, but not structure type,

was associated with NO_2 concentrations in the heating period for homes with unvented heaters (76.2 ± 1.4 ppb versus 55.9 ± 3.9 ppb in homes with aluminum and wood windows, respectively), but not in homes with vented heaters. In the non-heating season, mean NO_2 levels in the home varied by type of structure (steel/concrete or wood) and type of window frames (aluminum or wood), with wood structures and frames indicating a less airtight dwelling.

AX3.5 PERSONAL EXPOSURE

Components of Personal Exposure

Human exposure to NO_2 consists of contact at the air boundary layer between the human and the environment at a specific concentration for a specified period of time. People spend various amount of time in different microenvironments with various NO_2 concentrations. The integrated NO_2 exposure is the sum of the individual NO_2 exposures over all possible time intervals for all environments. Therefore, the assessment of human exposures to NO_2 can be represented by the following equation:

$$E_T = \sum_{i=1}^{n} C_i f_i \qquad \text{(AX3-12)}$$

where E_T is the time-weighted personal exposure concentration over a certain period of time, n is the total number of environments that a person encounters, f_i is the fraction of time spent in the ith environment, and C_i is the average NO_2 concentration in the ith environment during the time fraction f_i. Depending upon the time fraction and environmental concentration we consider during exposure assessment, the exposure a person experiences can be classified into instantaneous exposure, peak exposure, averaged exposure, or integrated exposure. These distinctions are important because health effects caused by long-term low-level exposures may be different from those resulting from short-term peak exposures.

The equation above represents the average personal exposure concentration is a linear combination of the average concentration in the ambient environment and each microenvironment, weighted by an individual's fraction of time spent in that environment. Hence, personal exposure to NO_2 is influenced by the microenvironmental concentration and the amount of time spent in each microenvironment. In theory, a microenvironment could be any

three-dimensional space having a volume in which people spend a certain amount of time. In practice, microenvironments typically used to determine NO_2 exposures include residential indoor environment, other indoor locations, near-traffic outdoor environment, other outdoor locations, and in-vehicles. In other words, total personal exposure to NO_2 can be decomposed into exposure to NO_2 in different environments. An individual's total exposure (E_T) can also be represented by the following equation:

$$E_T = E_a + E_{nona} = \{y_o + \sum_i y_i[P_i a_i/(a_i + k_i)]\} C_a + E_{nona} = \{y_o + \sum_i y_i F_{inf_i}\} C_a + E_{nona} \quad \text{(AX3-13)}$$

subject to the constraint

$$y_o + \sum_i y_i = 1 \quad \text{(AX3-14)}$$

where E_a is the person's exposure to pollutants of ambient origin; E_{nona} is the person's exposure to pollutants that are not of ambient origin; y_o is the fraction of time people spend outdoors and y_i is the fraction of time they spend in microenvironment i; F_{infi}, P_i, a_i, and k_i are the infiltration factor, penetration coefficient, air exchange rate, and decay rate for microenvironment i.

In the case where microenvironmental exposures are dominated by one microenvironment, Equation AX3-13 may be approximated by

$$E_T = E_a + E_{nona} + \{y + (1-y)[Pa/(a + k)]\} C_a + E_{nona} = \alpha C_a + E_{nona} \quad \text{(AX3-15)}$$

where E_t is the total personal exposure, E_a is the exposure to ambient generated pollutants, E_{nonag} is the nonambient generated pollutants, and y is the time fraction people spent outdoors. Other symbols have the same definitions in Equation AX3-13. If microenvironmental concentrations are considered, then Equation AX3-15 can be recast as

$$C_{me} = C_a + C_{nona} = [Pa/(a + k)]C_a + S/[V(a + k)] \quad \text{(AX3-16)}$$

where C_{me} is the concentration in a microenvironment; C_a and C_{nona} the contributions to C_{me} from ambient and nonambient sources; S is the microenvironmental source strength; V is the volume of the microenvironment, and the symbols in brackets have the same meaning as in Equation AX3-15. In this equation, it is assumed that microenvironments do not exchange air with each other, but only with ambient air.

The NO_2 concentration in each microenvironment can show substantial spatial and temporal variability, which is determined by many factors, such as season, day of the week, personal age, occupation, house characteristics, personal activities, source emission rate, air exchange rate, and transport and removal mechanisms of NO_2. Failure to disaggregate total human exposure and assess human exposure in various microenvironments may result in exposure misclassification, which may obscure the true relations between ambient air pollution and health outcomes.

Studies reviewed in this section were generally conducted in North America (Canada, the United States, and Mexico) and European countries. Studies conducted in other parts of the world were not the primary focus of this science review because exposure patterns may not be similar to those in the United States. However, studies which might support general conclusions (not country or cultural specific conclusions) about NO_2 exposures will be included.

Either Palmes tubes or Yanagisawa badges or Ogawa samplers were used to measure personal exposures in most of the reviewed studies, and sometimes residential indoor and outdoor concentrations. Sampling time for each cartridge varied from 8 h to two weeks, and the study design covered (1) longitudinal, in which each subject is measured for many days; (2) pooled, in which each subject is measured for only one or two days, different days for different subjects; and (3) daily-average, in which many subjects are measured on the same day. Most studies focused primarily on children, and in some studies adults or people with respiratory diseases were taken as study population.

AX3.5.1 Personal Exposures and Ambient (Outdoor) Concentrations

Numerous epidemiological studies have shown a positive association between ambient (outdoor) NO_2 concentrations and adverse health effects. Since a causal association requires exposure, it is very important to evaluate personal exposure to ambient (outdoor) generated NO_2. In this section, topics related to the total personal exposure and ambient (outdoor) generated NO_2 will be evaluated, such as the levels of personal exposure and ambient (outdoor) NO_2, the attenuation factor of personal exposure to NO_2, the correlation between personal and ambient (outdoor) NO_2, and the factors determining the associations between personal exposure and ambient (outdoor) level. Based on the science review, the following key questions will be addressed: 1) When, where, how and how much are people exposed to ambient (outdoor)

generated NO_2? and 2) Is ambient (outdoor) NO_2 a good surrogate for personal total exposure or personal exposure to ambient (outdoor) NO_2?

Personal exposures in most of the studies considered here were less than the corresponding outdoor or ambient concentrations. In the presence of local sources (indoor or local traffic sources), personal exposure levels could be higher than outdoor or ambient levels (Spengler et al., 1994; Nakai et al., 1995; Linn et al., 1996; Spengler et al., 1996; Raaschou-Nielsen et al., 1997; Alm et al., 1998; Levy et al., 1998a; Monn et al., 1998; Liard et al., 1999; Krämer et al., 2000; Linaker et al., 2000; Mukala et al., 2000; Gauvin et al., 2001; Monn, 2001; Rotko et al., 2001; Sarnat et al., 2001; Kodama et al., 2002; Mosqueron et al., 2002; Ramirez-Aguilar et al., 2002; Rojas-Bracho et al., 2002; Lai et al., 2004; Nerriere et al., 2005; Sarnat et al., 2005; Sørensen et al., 2005; Kim et al., 2006; Sarnat et al., 2006).

In a probability based population exposure study in Los Angeles Basin, 48 h indoor, outdoor and personal exposures (pooled exposures) were reported for 682 participants (Spengler et al., 1994). Spengler et al. (1994) found that the median personal exposure was 35 ppb and the median outdoor level was 36 ppb. Linn et al. (1996) reported the results of a personal exposure study for 269 school children from three Southern California communities. During this longitudinal study, 24 h averaged personal exposures, as well as inside school, outside school and ambient central site NO_2 levels, were measured by Yanagisawa badges for one week for each season from 1992 to 1994. Results showed that mean personal exposure was 22 ppb and the mean central site concentration was 37 ppb. Kim et al. (2006) conducted a longitudinal, multi-pollutant exposure study in Toronto, Canada. During the study, personal exposures (24-h integrated by Ogawa sampler) to $PM_{2.5}$, NO_2 and CO were measured for 28 subjects with coronary artery disease one day a week for a maximum of 10 weeks, and were compared with ambient fixed site measurements. The mean NO_2 personal exposure was 14.4 ppb, which was lower than the ambient site concentrations (20-26 ppb). Sarnat et al. (2001) and Sarnat et al. (2005) reported multi-pollutant exposure studies in Baltimore and Boston. In the Baltimore study, 24 h averaged personal exposure and ambient $PM_{2.5}$, O_3, NO_2, SO_2, and CO were measured for 56 subjects (20 older adults, 21 children and 15 individuals with COPD) in the summer of 1998 and the winter of 1999. All subjects were monitored for 12 or 8 consecutive days in each of the one or two seasons. Median ambient NO_2 levels were higher than the median personal levels in both seasons (about 10 ppb in difference). During the winter, both ambient

and personal exposure to NO_2 were higher than the summer, the difference between ambient and personal exposure in winter was 1 to 2 ppb smaller than the difference in the summer. In the Boston study, 24-h averaged personal and ambient $PM_{2.5}$, O_3, NO_2, and SO_2 were measured for 20 healthy seniors and 23 schoolchildren. All subjects were measured for 12 consecutive 24-h periods in each of the 1 or 2 seasons. Ambient NO_2 levels were on average 6 to 20 ppb higher than the personal exposure levels for seniors during all sampling sessions. For children's exposure, ambient NO_2 levels were 7 to 13 ppb higher than the personal exposures in 4 out of 6 sampling sessions, and in the other two sampling sessions (one in summer and one in winter) ambient levels were 1.8 to 2.6 ppb lower than personal exposures. Sarnat et al. (2006) measured 24-h averaged ambient and personal $PM_{2.5}$, sulfate, elemental carbon, O_3, and SO_2 for 10 non-smoking seniors in Steubenville, Ohio during the summer and fall of 2000. For each subject, two consecutive 24 h personal exposure measurements were collected during each week for 23 weeks. Data were stratified by the presence of gas stoves in homes. Personal exposure was lower than the ambient level for homes without gas stoves (9.0 ppb for personal exposure versus 9.5 ppb for ambient level during the summer and 9.9 ppb versus 11.3 ppb during the fall), and higher than ambient levels for homes with gas stoves (12.3 ppb for personal exposure versus 9.5 ppb for ambient level during the summer and 15.7 ppb versus 11.3 ppb during the fall).

Nerriere et al. (2005) investigated factors determining the discrepancies between personal exposure and ambient levels in the Genotox ER study. During the study, forty-eight h averaged $PM_{2.5}$, PM_{10}, and NO_2 were collected in both summer and winter for each person in a cohort, with 60 to 90 nonsmoking volunteers composed of two groups of equal size for adults and children at four metropolitan areas in France (Grenoble, Paris, Rouen, and Strasbourg). In each city, subjects were selected so as to live in three different urban sectors contrasted in terms of air pollution: one highly exposed to traffic emissions, one influenced by local industrial sources, and a background urban environment. In each urban sector, a fixed ambient air monitoring station was used to simultaneously collect the same air pollutants as personal exposure samplers. Factors affecting the concentration discrepancies between personal exposure and corresponding ambient monitoring site were investigated by a multiple linear regression model. Results showed that the discrepancies were season, city and land use dependent. During the winter, city and land use can interpret 31% of the variation of the discrepancy, and during the summer 54% of the variation in the discrepancy can be interpreted by those factors. In most cases, ambient

concentrations were higher than the corresponding personal exposures. When using the ambient site to represent ambient levels, the largest difference between ambient and personal exposure was found at the "proximity to traffic" site, while the smallest difference was found at the "background" site. When using urban background site as ambient level, the largest difference was observed at the "industry" site, and the smallest difference was observed at the background site, which reflected the heterogeneous distribution of NO_2 in an urban area. During winter, differences between ambient site and personal exposure were larger than those in the summer. Age was not found to be a significant factor interpreting the discrepancies between ambient level and personal exposure.

Sørensen et al. (2005) reported that during the cold season, median personal exposure was higher than residential indoor and urban background concentrations, but lower than the residential outdoor and street station concentrations (designed to capture the close to traffic exposure). During the warm season, personal exposure was again lower than the street station concentration but higher than the residential indoor, outdoor, and urban background concentrations. The implication of these findings is that ambient concentrations are the primary factor in determining exposures when there is no or little contribution from indoor sources and that traffic is the most significant NO_2 source in this study.

The relative levels of ambient and personal exposure can also be expressed as ratios of personal/ambient (Levy et al., 1998a; Rojas-Bracho et al., 2002; Sarnat et al., 2006). As shown in Equation AX3-15, personal exposure is related to ambient concentration through the infiltration factor, the fraction of time people spend outdoors, indoor sources and outdoor concentration. In the absence of indoor sources, the ratio of personal exposure to ambient concentration is sometimes also called the attenuation factor (α), which is always less than or equal to one, and it is a function of infiltration factor (F_{inf}) and the fraction of time people spend outdoors (y). The attenuation factor can be derived directly from measured personal and outdoor concentrations or calculated from measured or estimated values of the parameters a, k, and P (see Equation AX3-13 and Equation AX3-15) and the time spent in various microenvironments from activity pattern diaries (Wilson et al., 2000). Because α depends on building and lifestyle factors, air exchange rate, and NO_2 decay rate, it will vary to a certain extent from region-to-region, season-to-season, and by the type of indoor microenvironment. Consequently, predicted exposures based on these physical modeling concepts provide exposure distributions derived

conceptually as resulting from building, lifestyles, and meteorological considerations. For any given population, the distribution of the coefficient α may represent substantial intra- and inter-personal variability based on personal activity patterns, building and other microenvironmental characteristics, and proximity to ambient and indoor sources. Distributions of α should be determined using population studies in order to evaluate the uncertainty and variability associated with model exposures. Unfortunately, only a few studies have reported the value and distribution of the ratio of personal to ambient, and even fewer studies reported the value and distribution of attenuation factors based on sophisticated study designs. Rojas-Bracho et al. (2002) reported the median personal/outdoor ratio was 0.64 (with an interquartile range (IQR) of 0.45). Although it was less than one, the authors also reported the indoor/outdoor ratio (0.95 with an IQR of 0.48) of NO_2 and based on the indoor/outdoor ratio, the authors pointed out that the high median indoor/outdoor ratio was greater than the estimated effective penetration efficiency, which supports the argument of the importance of indoor sources to indoor NO_2 levels. Therefore, the attenuation factor in this study should be smaller than the ratio of personal/ambient, which was 0.64. Sarnat et al. (2006) reported that the ratio of personal/ambient for NO_2 was 2.05 and 1.27 for subjects with and without gas stoves in their homes. The large personal/ambient ratio for the latter might be attributed to the influence of indoor or local sources that were not identified and/or partly to measurement error.

The attenuation factor is one of the keys to evaluate personal exposure to ambient generated NO_2, or ambient contribution to personal exposure. However, the ratio of personal exposure/ambient concentration will not accurately reflect the attenuation factor in the presence of indoor sources. As shown above, in many cases, the ratio of personal exposure and ambient concentration was above one, which is physically impossible for the attenuation factor. The random component superposition (RCS) model is an alternative way to calculate attenuation factor using observed ambient and personal exposure concentrations (Ott et al., 2000). The Random Component Superposition (RCS) statistical model (shown in Equation AX3-15) uses the slope of the regression line of personal concentration on the ambient or outdoor NO_2 concentration to estimate the population average attenuation factor and means and distributions of ambient/outdoor and nonambient contributions to personal NO_2 concentrations (the intercept of the regression is the averaged nonambient contribution to personal exposure). This model

assumes a linear superposition of the ambient and nonambient components of exposure and lack of correlation between these two components.

The RCS model derives a mean α across all homes (assuming the infiltration behavior and time budget for all people are the same) from the linear regression of measured values of E_t on C_a. The product of the constant α and C_a from each home provides an estimate of the mean and distribution of E_a for the population of study homes. In practice, the mean and distribution of nonambient contributions (E_{nona}) are given by the difference, E_t - E_a, on a home-by-home basis. The RCS-predicted distribution of E_a across the population of study homes is given by the product of the constant α and C_a from each home, and the mean of the ambient contribution is the difference between the mean total personal exposure and the intercept of the regression line. The RCS model has been widely applied to PM exposure studies PTEAM, THEES, Toronto, and RIOPA studies (Ott et al., 2000; Meng et al., 2005), but researchers have not intentionally used this model for NO_2 exposure assessments. Although many studies explored the relationship between personal exposure and ambient NO_2 concentrations using regression models, most of those studies are not useful for evaluating the attenuation factor or helping answer the question of how much personal NO_2 exposure comes from ambient air, either because only R^2 was reported, or because log-transformed concentrations were used in the regression model, or because physically meaningless multiple linear regression models (exploratory variables were not independent of each other, e.g., both indoor, outdoor, indoor sources from questionnaire responses and air exchange rate were used as exploratory variables) were used to interpret personal exposure variations. Only those simple linear regression models (personal versus ambient or personal versus outdoor) and physically meaningful multiple linear regression models (personal versus ambient + indoor source measured or identified by questionnaire) are useful for evaluating the attenuation factor, and those models are summarized in Table AX3.11. The intercept of the regressions (i.e., the nonambient contribution to personal exposure) varies widely from study to study (5 ppb to 18 ppb) and thus depends strongly on time and location. The slope of these regression models (i.e., the population average attenuation factor) varies between 0.3 to 0.6 in most of the studies. The attenuation factor is determined by air exchange rate, penetration and decay rate of NO_2 and also the fraction of time people spend outdoors. Sørensen et al. (2005) found that the attenuation factor was larger in the summer than in the winter. However, Sarnat et al. (2006) found opposing results and said the reason was unknown. Based on the

regression model and reported mean personal exposure values, the ambient and nonambient contribution to personal exposure could be calculated using the method described above. Since most researchers did not report the mean personal exposure and the regression model at the same time, ambient and nonambient contributions can only be calculated in four studies as shown in Table AX3.12. The ambient contribution to population exposures varies from 20% to 50% in these four studies.

The RCS model calculates ambient contributions to indoor concentrations and personal exposures based on the statistical inferences of regression analysis. However, personal-outdoor regressions could be affected by extreme values (outliers either on the x or the y axis), such as a high nonambient exposure on a day with low ambient concentration or vice versa. For this reason outliers must be identified and their influence on the infiltration factor or attenuation factor in the RCS model must be evaluated in order to obtain a robust result. Another limitation of the RCS model is that this model is not designed to estimate ambient and nonambient contributions for individuals, in part because the use of a single attenuation factor does not account from the large home-to home variations in actual air exchange rates, and penetration and decay rates of NO_2. As suggested by Meng et al. (2005) the use of a fixed attenuation factor might underestimate ambient contributions to indoor concentrations and personal exposures and could also overlook some of the exposure errors and cause large uncertainties in risk estimates.

The estimation of the ambient and nonambient contribution to personal exposure could be improved by allowing for variations in air exchange rate, penetration and decay rate of NO_2, and the variations in the fraction of time people spend outdoors. The mass balance model described in Equation AX3-15 gives more flexibility than the RCS model if the distributions of P, k, a, and y are known. A comprehensive assessment of the impact of ambient sources on personal exposure would require detailed consideration of the mechanisms of NO_2 formation, transformation, transport and decay. In the research field of NO_2 exposure assessment, no published reports were found that use the mass balance model to explore the relationship of personal exposures to ambient NO_2 concentrations. As mentioned in Section 3.4.2, the only reported k values were 0.99 h-1 by Yamanaka (1984), and people always assumes the penetration coefficient (P) is one for NO_2, which might overestimate the ambient contribution due to the chemical reactivity of NO_2 during penetration.

The association between personal exposure and ambient NO_2 was quantified by Pearson correlation coefficient (r_p), Spearman correlation coefficient (r_s), or coefficient of determination (R^2) in regression models (Spengler et al., 1994; Linn et al., 1996; Spengler et al., 1996; Raaschou-Nielsen et al., 1997; Alm et al., 1998; Levy et al., 1998a; Monn et al., 1998; Liard et al., 1999; Krämer et al., 2000; Linaker et al., 2000; Mukala et al., 2000; Gauvin et al., 2001; Monn, 2001; Rotko et al., 2001; Sarnat et al., 2001; Kodama et al., 2002; Rojas-Bracho et al., 2002; Lai et al., 2004; Sarnat et al., 2005; Kim et al., 2006; Sarnat et al., 2006). In Table AX3.13, the associations between personal exposure and ambient concentration found in these studies are summarized.

The association between personal NO_2 exposure and ambient/outdoor NO_2 concentration varied from poor to good as shown in Table AX3.13. The strength of the correlation between personal exposure and ambient/outdoor concentration for a population is determined by the variations in indoor or other local sources, air exchange rate, penetration and decay rate of NO_2 in different microenvironment, and time people spend in different microenvironments with different NO_2 concentrations. The relationship is also a function of season and location (rural/urban). Alm et al. (1998) indicated that the association between personal exposure and outdoor concentration was stronger than the correlation between personal exposure and central site concentration. However, Kim et al. (2006) pointed out that the association was not improved using the ambient sampler closest to a home. Home ventilation is another important factor modifying the personal-ambient relationships; we expect to observe the strongest associations for subjects spending time indoors with open windows. Alm et al. (1998) and Kodama et al. (2002) observed the association between personal exposure and ambient concentration became stronger during the summer than the winter. However, Sarnat et al. (2006) reported that R^2 decreased from 0.34 for low ventilation population to 0.16 for high ventilation population in the summer, and from 0.47 to 0.34 in the fall. This might be a caution that the association between personal exposure and ambient concentration is complicated and is determined by many factors. Exposure misclassification might happen if a single factor, such as season or ventilation status, is used as an exposure indicator. Another factor affecting the personal to ambient association is the subject's location, with higher correlation for subjects living in the rural areas and lower correlation with subjects living in the urban areas (Rojas-Bracho et al., 2002; Alm et al., 1998). Spengler et al. (1994) also observed that the relationship between personal exposure and outdoor

concentration was highest in areas with lower ambient NO_2 levels ($R^2 = 0.47$) and lowest in areas with higher ambient NO_2 levels ($R^2 = 0.33$). This might reflect the highly heterogeneous distribution, or the effect of local sources of NO_2 in an urban area, and personal activities are more diverse in an urban area. However, this factor (location: urban vs. rural) might also interact with indoor sources because indoor sources could explain more personal exposure when ambient concentrations become lower and more homogeneously distributed.

The association is also affected by indoor or local sources, and the association becomes stronger after those sources are controlled in the model. Raaschou-Nielsen et al. (1997) observed that R^2 increased from 0.15 for general population to 0.49 for a population who spent less than 2% of their time close to gas appliances and passive smoking in Copenhagen urban area, and R^2 increased from 0.35 to 0.45 in the rural area for the population with the same characteristics. When those who reported exposure to either gas appliances or passive smoking were excluded, R^2 increased to 0.59 in urban and 0.46 in the rural districts. Spengler et al. (1994) observed that less of the variation in personal exposure was explained by outdoor concentrations for those who had gas ranges with pilot lights ($R^2 = 0.44$) than it is for the other two groups ($R^2 = 0.52$). When there is little or no contribution from indoor sources, ambient concentrations are the primary factor in determining exposure, but if there are continuous indoor sources, the influence of outdoor levels decrease. In the VESTA study, Gauvin et al. (2001) reported low R^2s in all three cities. R^2s increased for all three cities after controlling indoor air sources (e.g., gas cooking) and ambient traffic densities: R^2 increased from 0.01 to 0.43 for Grenoble, from 0.04 to 0.50 for Toulouse, and from 0.02 to 0.37 for Paris. Other factors, such as cross-sectional vs. longitudinal study design, and sampling duration might also affect the strength of the association. However, the current science review cannot give a clear picture of the effects by those factors due the lack of key studies and data.

The correlation coefficient between personal exposure and ambient/outdoor concentration has different meanings for different study designs. There are three types of correlations generated from different study designs: longitudinal, "pooled," and daily-average correlations. Longitudinal correlations are calculated when data from a study includes measurements over multiple days for each subject (longitudinal study design). Longitudinal correlations describe the temporal relationship between daily personal NO_2 exposure or microenvironment concentration and daily ambient NO_2 concentration for each individual subject. The longitudinal correlation

coefficient may differ for each subject. The distribution of correlations across a population could be obtained with this type of data. Pooled correlations are calculated when a study involves one or only a few measurements per subject and when different subjects are studied on subsequent days. Pooled correlations combine individual subject/individual day data for the calculation of correlations. Pooled correlations describe the relationship between daily personal NO_2 exposure and daily ambient NO_2 concentration across all subjects in the study. Daily-average correlations are calculated by averaging exposure across subjects for each day. Daily-average correlations then describe the relationship between the daily average exposure and daily ambient NO_2 concentration.

The type of correlation analysis can have a substantial effect on the value of the resultant correlation coefficient. Mage et al. (1999) mathematically demonstrated that very low correlations between personal exposure and ambient concentrations could be obtained when people with very different nonambient exposures are pooled, even though their individual longitudinal correlations are high. Data shown in Table AX3.13 demonstrate that the longitudinal correlations between personal exposure and ambient NO_2 concentrations were higher than the correlations obtained from a pooled data set.

In conclusion, personal exposure to ambient/outdoor NO_2 is determined by many factors. Physically, the determinant factors are ambient concentration, air exchange rate, NO_2 penetration and decay rate, and also the fraction of time people spend outdoors. These factors are in turn determined by factors, such as season, location of home, outdoor temperature and so on. These factors all help determine the contribution of ambient/outdoor generated NO_2 to personal exposures. Personal activities determine when, where and how people are exposed to NO_2. The variations of these physical factors and indoor sources determine the strength of the association between personal exposure and ambient concentrations both longitudinally and cross-sectionally. In the absence of indoor and local sources, the personal exposure level is in between the ambient level and the indoor level, but in the presence of indoor and local sources, personal exposures could be much higher than both indoor and outdoor concentrations. Again, the discrepancies between personal exposures and ambient levels are determined by the considerations given above. Most researchers found that personal NO_2 was significantly associated with ambient NO_2 but the strength of the association ranged from poor to good. Based on that finding, some researchers concluded that ambient NO_2 is a good surrogate for personal exposure, while others

reminded us that caution must be exercised if ambient NO_2 is used as a surrogate for personal exposure. The crude association between personal exposure and ambient monitors could be improved when indoor or other local sources are well controlled during exposure assessment. The ambient contribution to personal exposure could be evaluated by the attenuation factor, which is the ratio of personal exposure to ambient level in the absence of indoor sources, or the slope of the RCS regression model. The attenuation factor in the studies shown in Table AX3.11 ranged from 0.3 to 0.6. The ambient and nonambients contributions could also be calculated from the RCS model, although only a few studies provide enough information for us to calculate them. The accuracy and precision of the estimation of ambient and nonambient contributions to personal exposures could be improved if the variations for the physical factors given above were known. The mass balance model could give a more accurate and precise estimation if we knew the distributions of these key physical factors.

Because people are exposed to ambient NO_2 in microenvironments, and the fact that NO_2 is heterogeneously distributed in urban areas (as shown in Section AX3.3.2), the association of personal exposure to ambient NO_2 could be modified by microenvironmental characteristics. Personal total exposure will be decomposed and further evaluated in each microenvironment in the following section.

AX3.5.2 Personal Exposure in Microenvironments

Personal Exposure in the Residential Indoor Environment

People spend most of their daily time in a residential indoor environment (Klepeis et al., 2001). NO_2 found in an indoor environment originates both indoor and outdoors; and therefore, people in an indoor environment are exposed to both indoor and outdoor generated NO_2. The physical parameters, which determine personal exposure to ambient and nonambient generated NO_2, have been shown in Equations AX3-13 to AX3-16. In a residential indoor environment, personal exposure to NO_2 can be summarized below (notations are the same as those in Equations AX3-13 to AX3-16):

$$E_t = E_a + E_{nona} = \alpha C_a + E_{nona} = \{y + (1-y)[Pa/(a+k)]\}C_a + E_{nona} = \{y + (1-y)F_{inf}\}C_a + E_{nona} \quad \text{(AX3-17)}$$

if people spend 100% of their time indoors, the equation above can be recast as

$$E_t = E_a + E_{nona} = \alpha C_a + E_{nona} = F_{inf} C_a + E_{nona} = [Pa/(a+k)]C_a + S/[V(a+k)] = C_a + C_{nona} \quad \text{(AX3-18)}$$

In other words, in a residential indoor environment, personal exposure concentration equals the residential indoor concentration (if there is no personal cloud) which can be broken down into two parts: indoor generation and ambient contribution.

In a residential indoor environment, the relationship between personal NO_2 exposure and ambient NO_2 can be modified by the indoor environment in the following ways: (1) during the infiltration processes, ambient NO_2 can be lost through penetration and decay (chemical and physical processes) in the indoor environment, and therefore, the concentration of indoor NO_2 of ambient origin is not the ambient NO_2 concentration but the product of the ambient NO_2 concentration and the infiltration factor (F_{inf}, or α if people spend 100% of their time indoors); (2) in an indoor environment, people are exposed to not only ambient generated NO_2 but also indoor generated NO_2, and therefore, the relative contribution of ambient and nonambient NO_2 to personal exposure depends not only on the ambient NO_2 concentration but also on the infiltration factor (attenuation factor) and the indoor source contribution; (3) the strength of the association between personal exposure to NO_2 of ambient origin and ambient NO_2 concentration is determined by the temporal and spatial variation in the infiltration factor; and (4) the strength of the association between personal total exposure and ambient NO_2 is determined by the variation in the indoor source contribution and the variation in the infiltration factor. Below, factors affecting infiltration factor and the indoor source contribution will be evaluated, and the key issues, such as those mentioned above, related to ambient contribution to personal NO_2 exposure will be addressed.

Infiltration factor (F_{inf}) of NO_2, the physical meaning of which is the fraction of ambient NO_2 found in the indoor environment, is determined by the NO_2 penetration coefficient (P), air exchange rate (a), and the NO_2 decay rate (k), through the equation $F_{inf} = Pa/(a + k)$. Information on P and k for NO_2 is sparse. In most mass balance modeling work, researchers assume P equals 1 because NO_2 is a gas, and assume k equals 0.99 h^{-1}, which is cited from Yamanaka (1984). Yamanaka (1984) systematically studied the decay rates of NO_2 in a typical Japanese living room. The author used a chemical luminescence method to monitor the decay process of indoor-originated NO_2. The author observed that the decay process of NO_2 followed approximately first-order kinetics. The author also pointed out that the NO_2 decay processes was

both surface type and relative humidity (RH) dependent: Under low RH (43.5-50%), the sink rate of NO_2 was 0.99 ± 0.19 h^{-1}, independent of interior surface properties; however, the NO_2 decay rate increased in proportion to RH above 50%, and in that RH range, the decay rate depended on the interior surface properties. Yang et al. (2004) estimated a decay rate of 0.94 h^{-1} for Seoul and 1.05 h^{-1} for Brisbane. As it is well known, the decay rate is dependent on lots of indoor parameters, such as indoor temperature, relative humidity, surface properties, surface-to-volume ratio, the turbulence of air flow, and co-existing pollutants, et al. However, in the indoor air modeling studies, a decay rate of 0.99 h^{-1} is a widely accepted parameter (Dimitroulopoulou et al., 2001; Kulkarni et al., 2002). As a result, it will over- or underestimate the real NO_2 decay rate. A penetration coefficient (P) of 1 is also widely accepted for NO_2 (Kulkarni et al., 2002; Yang et al., 2004). No systematic investigations have been found on NO_2 penetration behaviors. As a general principle, the upper limit of the penetration coefficient is 1, and it would be less than 1 if NO_2 lost during penetration due to diffusion and chemical reactions. Therefore, using a penetration coefficient of 1 gives an upper bound to the estimated infiltration coefficient. Among P, k, and a, air exchange rate (a) is the most solidly based parameter and can be obtained from a nationwide database (Pandian et al., 1998).

Although specific P, k, and a were not reported by most studies, a number of studies investigated factors affecting P, k, and a (or indicators of P, k, and a), and their effects on indoor and personal exposures (Lee et al., 1996; Cotterill et al., 1997; Monn et al., 1998; García-Algar et al., 2003; Sørensen et al., 2005; Zota et al., 2005). García-Algar et al. (2003) observed that double-glazed windows had significant effect on indoor NO_2 concentrations. Homes with double-glazed windows had lower indoor concentrations (6 ppb lower) than homes with single glazed windows. Cotterill et al. (1997) reported that single or double glazed window was a significant factor affecting NO_2 concentrations in kitchen in the gas-cooker homes (31.4 ppb and 39.8 ppb for homes with single and double glazed windows, respectively). The reduction of ventilation can block outdoor NO_2 from coming into the indoor environment, and at the same time it can also increase the accumulation of indoor generated NO_2. The same effect was found for homes using air conditioners. Lee et al. (2002) observed that NO_2 was 9 ppb higher in homes with an air conditioner than homes without. The authors also observed that the use of humidifier would reduce indoor NO_2 by 6 ppb. House type was another factor reported affecting ventilation (Lee et al., 1996; García-Algar et al., 2003). Lee et al. (1996) reported that the building type was

significantly associated with air exchange rate: the air exchange rate ranged from 1.04 h^{-1} for single dwelling unit to 2.26 h^{-1} for large multiple dwelling unit. Zota et al. (2005) reported that the air exchange rates were significantly lower in the heating season than the non-heating season (0.49 h^{-1} for the heating season and 0.85 h^{-1} for the non-heating season respectively). It should be pointed out that both *P* and *k* are functions of complicated mass transfer mechanisms on the indoor surfaces, and therefore they are associated with air exchange rate, which has an impact on the turbulence of air flows indoors. However, the relationship between *P*, *k*, and *a* has not been thoroughly investigated. Factors mentioned above can significantly affect *P*, *k*, and *a*, and thus affect the relationship between indoor and outdoor NO_2 concentration, and personal exposure and outdoor NO_2 concentration.

Due to the lack of specific *P*, *k*, and *a* for study homes or a study population, instead of using *P*, *k*, and *a*, alternative approaches to obtain the infiltration factor are the ratio of indoor/outdoor NO_2 and the regression based RCS model. The basic rationale of the RCS model has been introduced in the previous section. Without indoor sources, the ratio between indoor NO_2 and outdoor NO_2 should be always less than or equal to 1. If the indoor to outdoor ratio is larger than 1 (after adjusting for measurement error), we can surely say that indoor sources exist. However, if an indoor/outdoor ratio is less than one, we cannot exclude the effect of indoor sources; otherwise, the infiltration factor would be overestimated. In order to use an indoor/outdoor ratio as the infiltration factor, study designs and questionnaires must be carefully read, and only the ratio for homes without identified indoor sources can be used as an indicator of infiltration factor. The population averaged infiltration factor is the slope of the regression line of indoor concentration vs. outdoor concentration. The reliability of the regression slope is dependent upon the sample size and how to deal with the outlier effects. Indoor/outdoor ratios and the regression slopes are summarized in Table AX3.14. Those numbers, which can be considered as an infiltration factor, are underlined and marked with bold font. Most of the infiltration factors ranges from 0.4 to 0.7. Theoretically, infiltration factor is a function of air exchange rate, which has been indicated by season in some studies. However, most studies do not report the infiltration factor by season, and therefore, a seasonal trend of infiltration factor could not be observed in Table AX3.14.

As mentioned before, personal NO_2 exposure is not only affected by air infiltrating from outdoors but also by indoor sources. The NO_2 residential indoor sources reported are gas cooking, gas heating, kerosene heating, smoking and burning candles (Schwab et al., 1994; Spengler et al., 1994; Nakai et al., 1995; Lee et al., 1996; Linaker et al., 1996; Cotterill et al., 1997; Farrow et al., 1997; Kawamoto et al., 1997; Lee, 1997; Raaschou-Nielsen et al., 1997; Alm et al., 1998; Levy et al., 1998a; Monn et al., 1998; Garrett et al., 1999; Chao, 2001; Dennekamp et al., 2001; Dutton et al., 2001; Emenius et al., 2003; Kodama et al., 2002; Lee et al., 2002; Mosqueron et al., 2002; García-Algar et al., 2003; García-Algar et al., 2004; Lai et al., 2004; Lee et al., 2004; Yang et al., 2004; Zota et al., 2005; Sørensen et al., 2005; Lai et al., 2006). Spengler et al. (1994) reported that personal exposures in homes with gas range with pilot light were 15 ppb higher than those in homes with electric range, and it was 5 ppb higher in homes with gas range without pilot light than homes with electric ranges. Schwab et al. (1994) reported that homes with gas stove with pilot light had higher indoor NO_2 concentrations (peak concentrations ranging from 30 to 35 ppb), followed by homes with gas stove without a pilot light (peak concentrations ranging from 15 to 20 ppb) and then homes with electric stoves (peak concentrations ranging from 5 to 10 ppb). In an international study, Levy et al., (1998a) reported that the use of a gas stove in the home was the dominant activity influencing NO_2 concentrations with a 67% increase in mean personal NO_2 exposure and an increase in indoor-outdoor ratios from 0.7 to 1.2. Smoking was found to be another significant factor elevating personal and indoor NO_2 exposure. Monn et al. (1998) reported that during 1-week integrated measurement, smoking contributed 1 ppb more NO_2 exposure. Alm et al. (1998) reported that one-week integrated personal NO_2 exposure for smokers and nonsmokers were 12.9 ppb and 10.7 ppb, respectively. Zota et al. (2005) observed that smoking was not a significant indoor source. However, the authors pointed out that the effect of smoking might have been overwhelmed by the presence of the gas stove. Sørensen et al. (2005) found that burning candles were significantly associated with the elevation of indoor NO_2 ($p = 0.02$). NO_2 concentration in an indoor environment affected by the indoor sources is not homogeneously distributed: NO_2 concentration is usually the highest in the kitchen, lowest in the bedroom and the concentration in a livingroom is in between as shown in Table AX3.15. The concentration differences between a bedroom and a kitchen ranged from 1 ppb to 28 ppb, and largest difference occurred in homes with gas stoves.

The concentration differences in indoor microenvironments reflect the differences in personal exposure in those microenvironments, which is related to personal activities and behaviors. People who spend more time in a kitchen are expected to have higher NO_2 exposures. Also, in most exposure studies, integrated indoor and personal exposures were measured from 2 days to 2 weeks with passive samplers. Therefore, the peak exposure concentration could be even higher.

Indoor source contributions to indoor and personal exposure are determined by indoor source strength (S), house volume (V), air exchange rate (a) and the NO_2 decay rate (k) in an indoor environment, through the equation $C_{nona} = S/[V(a + k)]$. Indoor source strength has been summarized in a previous section (Indoor sources and concentrations of nitrogen oxides). With a mass balance approach, Yang et al. (2004) reported that the source strength for electric range was 3.5 ppb/h, 11.5 ppb/h for gas range in Brisbane, and 23.4 ppb/h for gas range in Seoul. The age of house and the house type are associated with ventilation, indoor sources, and house volume. As mentioned before, Lee et al. (1996) reported that the building type was significantly associated with volume of dwelling unit, and air exchange rate. Garrett et al. (1999) reported that older houses were associated with higher nitrogen dioxide levels, possibly as a result of older and less efficient appliances in older homes or due to smaller rooms.

The relative contribution of indoor and outdoor NO_2 to personal and indoor exposures can be easily and precisely calculated if we know the physical determinants, such as P, k, a, and indoor source strength. Probability based exposure models, such as SHEDS and APEX, could be used to evaluate the personal exposure to indoor and outdoor generated NO_2. Basically, those exposure models incorporate the physical and chemical processes determining indoor pollutant concentrations as a function of outdoor concentration, indoor emission rates and building characteristics; the combination of a microenvironment model and personal activity model will allow researchers to evaluate the personal exposure to indoor and outdoor generated NO_2. Due to the lack of those parameters in publications, we are going to use a regression based RCS model to evaluate the contribution of indoor and outdoor generated NO_2 to personal exposure. The rationale to use the RCS model to estimate indoor and outdoor contribution to indoor and personal NO_2 have been introduced in the previous section. In summary, the regression intercept of indoor NO_2 concentration vs. outdoor NO_2 concentration is the population mean indoor contribution to indoor NO_2; and the difference between the population mean NO_2 and the

intercept in the population mean of outdoor contribution to indoor NO_2. The RCS model results are summarized in Table AX3.16. As shown in Table AX3.16, the overall ambient contribution to indoor NO_2 is around 70% with a wide range from 40 to 90%. Indoor generated NO_2 contribution is 10-20% less for homes with electric stoves if electric stove then indoor contribution is usually zero. With the lack of indoor sources, the role of indoor environment is a sink for outdoor generated NO_2 due to physical and chemical losses of NO_2 in the indoor environment (Yamanaka et al., 1984; Ekberg 1996; Kraenzmer 1999; Chao et al., 2001). Chao (2001) reported that the average sink strength of NO_2 in an indoor environment in Hong Kong was 0.42 mg/h.

In theory, personal exposure of ambient origin should be at least as much as the indoor NO_2 of ambient origin in that people spend time in either an indoor or an outdoor environment. However, it was shown in the previous part (Table AX3.12) that the ambient contribution to population exposure ranged from 20% to 50% based on four studies (Rojas-Bracho et al., 2002; Monn et al., 1998; Levy et al., 1998a; Spengler et al., 1994); and results here show that the ambient contribution to indoor NO_2 is around 70% with a wide range from 40 to 90% based on another four studies (Mosqueron et al., 2002; Yang et al., 2004; Kulkarni et al., 2002; Monn et al., 1998). It is not clear at present why the indoor NO_2 of ambient origin is larger than the personal NO_2 exposure of ambient origin.

The strength of the indoor, outdoor and personal NO_2 associations (r_p: Pearson correlation coefficient; r_s: Spearman correlation coefficient; and R^2: coefficient of determination) are summarized in Table AX3.17. The strength of the associations are determined by the variation in F_{inf} (*P*, *k*, and *a*) and indoor source contributions from home to home and from day to day. In general, the correlation between indoor and outdoor NO_2 ranges from poor to good (r_p: 0.06 to 0.86). When we break down the correlation coefficient by season and indoor sources, it is obvious that the association between indoor and outdoor NO_2 is stronger during spring and summer but weaker during wintertime, and the association is stronger for homes without indoor sources but weaker for homes with strong indoor sources. Mukala et al. (2000) reported an r_p of 0.86 for the indoor and outdoor NO_2 association during the spring and it reduced to 0.54 during the winter. Spengler et al. (1994) reported that the associations were 0.66 and 0.75 (r_p) for homes with and without air conditioning system, respectively. Emenius et al. (2003) reported that the association between indoor and outdoor NO_2 was 0.69 (r_p) for

homes without smoker and without gas stove using, but the association was not significant for homes with gas stove or smokers. Yang et al. (2004) reported that the indoor and outdoor NO_2 association was 0.70 (R^2) for homes with electric ranges, and was 0.57 (R^2) for homes with gas ranges. In other words, personal exposure to ambient NO_2 in a residential indoor environment will be modified the least when the air exchange rate is high and the indoor source contribution is not significant. Considering the large spatial variation in ambient NO_2 concentrations and the relative sparseness of ambient NO_2 monitors, the associations between indoor and outdoor concentrations are usually stronger than the associations between indoor and ambient concentrations. As shown in Table AX3.17, a stronger personal vs. residential indoor relationship than the personal vs. outdoor relationship has been reported by most studies (Lai et al., 2004; Monn et al., 1998, Levy et al., 1998a; Spengler et al., 1994; Kousa et al., 2001; Linaker et al., 1996), which is a reminder that personal exposure to ambient NO_2 mostly happens in the residential indoor environment. It should be pointed out that the association between indoor, outdoor and personal NO_2 and the relative contributions of indoor and outdoor NO_2 to indoor and personal exposures were calculated based on time integrated indoor, outdoor and personal NO_2 measurement with passive samplers and an average measurement time of a couple of days to two weeks. In most studies, an equilibrium condition was assumed and the effects of dynamics on the indoor, outdoor, and personal association were not evaluated, which could result in missing the peak exposure and obscuring the real short-term outdoor contribution to indoor and personal exposure. For example, the NO_2 concentrations at locations close to busy streets in urban environments may vary drastically with time. If the measurement is carried out during a non-steady-state period, the indoor/outdoor concentration ratio may indicate either a too low relative importance of indoor sources (if the outdoor concentration is in an increasing phase) or a too high relative importance of indoor resources (if the outdoor concentration is in a decreasing phase). The lower the air exchange rate, the greater the error due to the effects of transients (Ekberg et al., 1996).

School and Office

Workplaces (schools and offices) are the places where people spend most of their time after homes in an urban area. The location, indoor sources as well as the ventilation pattern of schools and offices could be different from people's homes. Therefore, personal exposure patterns in

schools and offices could be different from exposure patterns at home. However, NO_2 concentrations in schools and offices have only been measured in only a few exposure studies.

Most studies reported the personal exposure levels were lower than or equal to office NO_2 levels. Lai et al. (2004) reported that a cohort in Oxford spent 17.5% of their daily time in offices, and mean personal total NO_2 exposure was 15 ppb and 16.8 ppb for mean office concentrations. Mosqueron et al. (2002) reported Paris office worker exposure levels and no significant difference was found between personal total exposure (22.8 ppb) and NO_2 concentrations in office (23.5 ppb). Personal exposures in schools were studied in Helsinki, Southampton and Southern California. Alm et al. (1998) and Mukala et al. (2000) reported the personal exposure levels in Helsinki for pre-school children. They reported that median personal exposures were lower than the median NO_2 concentrations measured inside the day care center (13.1 ppb for personal exposure versus 18.8 ppb for inside day-care center for downtown winter; 14.7 ppb versus 24.1 ppb for downtown spring; 8.9 ppb versus 15.2 ppb for suburban winter; and 8.9 ppb versus 13.1 ppb for suburban spring). Linaker et al. (1996) found that the geometric mean of school children exposures (18.8 ppb) was higher than geometric means of the NO_2 concentrations in classrooms (8.4 to 14.1 ppb) in a study of children's exposures to NO_2 in Southampton, UK. A similar exposure pattern was found by Linn et al. (1996) during the Southern California school children exposure study. During the study, personal exposure (22 ppb) was higher than the NO_2 concentration inside school (16 ppb). NO_2 concentration in school/office is determined by ambient NO_2 level, local traffic sources, floor height and building ventilation pattern. Partti-Pellinen et al. (2000) studied the effect of ventilation and air filtration systems on indoor air quality in a children's day-car center in Finland. Without filtration, NO_x and PM generated by nearby motor traffic penetrated readily indoors. With chemical filtration, 50 to 70% of nitrogen oxides could be removed. The authors suggested that the possible adverse health effects of nitrogen oxides and particles indoors could be countered by efficient filtration. Mosqueron et al. (2002) reported 24% of variations in in-office NO_2 concentrations could be explained by outdoor NO_2 levels (18%), and floor height (6%) and an inverse relation was observed between in-office concentration and floor height. Alm et al. (1998) attributed the high NO_2 concentration in the day-care center to its close to major roads. Obviously, the relative scale of personal exposure and school concentration also depends on personal activities outside schools and workplaces.

Significant associations between personal exposure and workplace concentrations were reported by most studies. Mosqueron et al. (2002) reported office NO_2 was a significant predictor of personal exposure and 15% of the personal exposure was explained by time weighted office NO_2 concentrations. Alm et al. (1998) reported population NO_2 exposures were highly correlated with the NO_2 levels inside the day-care centers ($R^2 = 0.88$). However, Lai et al. (2004) reported a nonsignificant Pearson correlation coefficient (0.15) between personal exposure and workplace indoor concentration and the authors suggested that the strong residential indoor sources and long time indoors obscured the personal versus office relationship. Personal total exposure is a function of NO_2 concentrations in different indoor and outdoor microenvironments and how long a person stays in that microenvironment. The large variation of NO_2 exposure in some microenvironments could obscure the association between personal exposure and NO_2 concentrations in other microenvironments.

In Traffic

On-road NO_2 concentrations could be substantially higher than ambient or residential outdoor NO_2 concentrations, especially in a street canyon, which are narrow with enclosing architecture and slow-moving traffic. As shown in Figure AX3.22, NO_2 in heavy traffic (~60 ppb) can be over twice the concentration in a residential outdoor level (~26 ppb) in North America (Lee et al., 2000). The UK and Scandinavian data in the plot may have been obtained outside homes close to traffic. Westerdahl et al. (2005) reported on-road NO_2 concentrations in Los Angeles ranging from 40 to 70 ppb on freeways, and 20 to 40 ppb on residential or arterial roads. People in traffic can potentially experience such high concentrations and NO_2 exposures due to the high air exchange rates for vehicles. Park et al. (1998) measured the air exchange rates in three stationary automobiles under four conditions: windows closed and no mechanical ventilation, windows closed with fan set on recirculation, windows open with no mechanical ventilation, and windows closed with the fan set on fresh air. The reported air exchange rates varied from 1.0 to 3.0 h^{-1} with windows closed and no mechanical ventilation to 36.2 to 47.5 h^{-1} with windows closed and the fan set on fresh air. It implies that the NO_2 concentration inside a vehicle is at least the same as the surrounding NO_2 concentration, or in other words, "on-road" NO_2 can quickly and almost completely infiltrate into the "in-vehicle" environment contribute to in-vehicle personal exposures. Although people only spend a small fraction of their time in

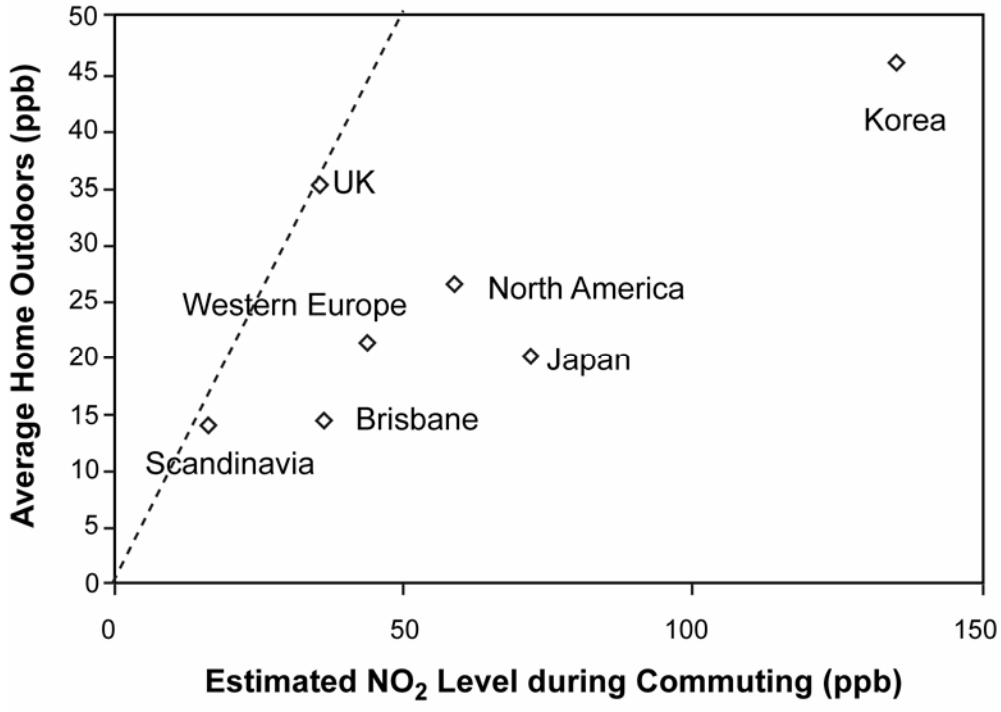

Figure AX3.22. Average residential outdoor concentration versus concentration during commuting for NO_2.

Source: Lee et al. (2000).

traffic (5% to 7%), exposure while commuting could be a significant contributor to personal exposure to NO_2 due to the high concentration of NO_2 in traffic. Liard et al. (1999) reported that both NO and NO_2 exposure levels increased with the number of hours spent in a car. During the study, NO and NO_2 concentrations were separated into three levels according to the distribution tertiles. Personal exposure levels increased from low to high when accordingly people spent from 2.5 h in a car to 6.7 h in a car. The same relationship only held for one of the two sampling periods, in which personal NO_2 exposures increased from low to high when the time people spent in a car increased from 3.5 h to 5.7 h.

Bell and Ashenden (1997) and Kirby et al. (1998) reported the NO_2 concentration along major roads and street canyons in UK, and they found that monthly mean NO_2 concentrations on major roads were consistently higher (up to 20 ppb) than those found 250 m away from the major roads. It is important to distinguish between short-term peak exposure and chronic

exposures because health effects associated with short-term peak exposures might be different from chronic exposures to ambient NO_2.

Other than infiltration of ambient air, the intrusion of the vehicle's own exhaust into the passenger cabin is another NO_2 source contributing to personal exposure while commuting. The intrusion of a school bus's own exhaust into the bus cabin was found by Sabin et al. (2005), but the fraction of air inside the bus cabin from the bus's own exhaust was small, ranging from 0.02% to 0.28%. Marshall and Behrentz (2005) also reported the intrusion of exhaust into the bus cabin and indicated that average per capita inhalation of emissions from any single bus is 10^5-10^6 times greater for a passenger on that school bus than for a typical resident in the same area. CARB (2007) reported that self-pollution increased with increasing age of the bus. Fuel type could be another factor affecting personal exposure while commuting. Son et al. (2004) found that the two-day averaged NO_2 exposures for taxi drivers using LPG fueled vehicles (26.3 ppb) were significantly lower than those using diesel-fueled vehicle (38.1 ppb). However, in another taxi driver exposure study, Lewné et al. (2006) did not find an effect on taxi driver exposures to NO_2 due to fuel differences (diesel versus petrol). Sabin et al. (2005) reported that NO_2 concentrations were significantly higher inside diesel buses than inside the compressed natural gas buses. CARB (2006) showed that the NO_2 concentrations on a conventional diesel bus was 2.8 times higher than the ambient concentration (76 ppb in cabin versus 27 ppb in ambient) while windows were closed, and 3.85 times higher than the ambient concentration (77 ppb in cabin versus 20 ppb in ambient) while windows were open. However, the ratio of cabin NO_2 to ambient NO_2 was much lower for a compressed natural gas bus: 1.2 for windows closed and 2.2 for windows opened.

While commuting, concentrations for personal exposure or in a vehicle cabin could be substantially higher than corresponding residential indoor, outdoor, and ambient concentrations. Sabin et al. (2005) measured concentrations of a number of pollutants (black carbon, particulate PAHs and NO_2 in school buses on routes in Los Angeles. Mean cabin concentrations for individual runs ranged from 24 to 120 ppb. Concentrations of NO_2 tended to be slightly higher for open compared to closed windows on urban routes. These concentrations were typically factors of 2.3 to 3.4 higher than at ambient monitors in the area. However, the highest ratios found ranged from 3.9 to 5.3. They concluded that children commuting in areas such as Los Angeles may be exposed to much higher levels of pollutants than are obtained at ambient, central

site monitors. Lewné et al. (2006) reported work hour exposures to NO_2 for taxi drivers (25.1 ppb), bus drivers (31.4 ppb) and lorry drivers (35.6 ppb). The ratios of in-vehicle exposures to urban background were 1.8, 2.7, and 2.8 for taxi drivers, bus drivers and lorry drivers, respectively. Due to the high peak exposures during commuting, total personal exposure could be underestimated if exposure in traffic are not considered; and sometimes exposure in traffic can dominate personal exposure to NO_2. In a personal exposure study in Brisbane and Queensland, Australia, two-day averaged indoor, outdoor, and personal NO_2 were measured by Yanagisawa badges (Lee et al., 2000). Lee et al. (2000) found that estimated personal exposures (22.5 ppb) significantly underestimated the measured personal exposures (28.8 ppb) if personal exposures in traffic were not considered. Son et al. (2004) reported two-day averaged indoor, outdoor, in vehicle and personal NO_2 concentrations measured by passive filter badges for 31 taxi drivers in Korea. Measured personal concentrations (30.3 ppb) were higher than both residential indoor (24.7 ppb) and residential outdoor concentrations (23.8 ppb). A stronger correlation was observed between personal NO_2 exposures and interior vehicle NO_2 levels, than for residential indoor and residential outdoor levels ($r_p = 0.89$ for Personal versus Vehicle, $r_p = 0.74$ for Personal versus Indoor; and $r_p = 0.71$ for Personal versus Outdoor).

Variations in traffic exposure could be attributed to time spent in traffic, type of vehicle, traffic congestion levels, encounters with other diesel vehicles, type of fuel and driving location (urban/rural) (Sabin et al., 2005; Son et al., 2004; Chan et al., 1999).

Microenvironments Close to NO_2 Sources

As suggested previously in this chapter, both large and small-scale variations exist in ambient NO_2 concentrations. In this section, those microenvironments and associated personal exposures, which are close to traffic sources and might make significant contributions to total personal NO_2 exposures are analyzed. These microenvironments could be residential outdoor environments and some other outdoor environments, such as parking lots and playgrounds; they could also be indoor environments as well, such as homes and classrooms. Concentrations in these microenvironments and personal exposure characteristics in these microenvironments will be summarized below.

Many studies show that outdoor NO_2 levels are strongly associated with distance from major roads (the closer to a major road, the higher the NO_2 concentration) (Gilbert et al., 2005; Roorda-Knape et al., 1998; Lal et al., 2001; Kodama et al., 2002; Gonzales et al., 2005; Cotterill

et al., 1997; Nakai et al., 1995). Meteorological factors (wind direction and wind speed), and traffic density are also important for interpreting measured NO_2 concentrations (Gilbert et al., 2005; Roorda-Knape et al., 1998; Rotko et al., 2001; Alm et al., 1998; Singer et al., 2004; Nakai et al., 1995). Gonzales et al. (2005) found an inverse correlation between NO_2 concentration and distance from a highway ($r_p = -0.81$, $p < 0.001$) in the El Paso region. Nakai et al. (1995) reported the results of a study designed to explore the differences of indoor, outdoor and personal exposure levels among residence zones located varying distances from major roads with heavy traffic in Tokyo. The authors found that outdoor NO_2 concentrations in Zone A (0-20 m from the road) was always the highest among the three zones (Zone B was 20-150 m from the road, and Zone C was a reference zone in a suburban area). The differences of the mean levels between Zone A and Zone C ranged from 11 ppb to 39 ppb. Kodama et al. (2002) reported NO_2 levels for indoor, outdoor and personal exposure among 150 junior high school student homes in two major traffic areas in Tokyo. Forty-eight h average NO_2 concentrations were measured by Yanagisawa badge. NO_2 tended to decrease according to distance from the roadside; the difference was about 10 ppb between the roadside (0-50 m) and the site far away from the road (200 m). Singer et al. (2004) reported results of the East Bay Children's Respiratory Health Study. The authors reported weekly integrated NO_2 and NO_x concentrations measured by Ogawa passive samplers placed outside ten elementary schools and selected student residences during 14 weeks in spring and 8 weeks in fall 2001. The authors found that NO_2 concentrations increased with decreasing downwind distance for school and neighborhood sites within 350 m downwind of a freeway, and schools located upwind or far downwind of freeways were generally indistinguishable from one another and regional pollution levels. An exponential equation was used to fit the measured concentrations to distance from the freeway: $C(x) = K_1 x^{K_2}$ where C is the measured concentration and x is the distance (m) from a freeway. A high R^2 was observed ($R^2 = 0.80$, $K_1 = 128$, and $K_2 = -0.356$ for NO_2; $R^2 = 0.76$, $K_1 = 376$, and $K_2 = -0.468$). According to this equation, NO_2 concentrations 100 m away from the freeway are about 20% of those at roadside.

Elevated NO_2 concentrations were also observed and reported in parking lots and school playgrounds. Lee et al. (1999) reported the concentration of NO_2 at a parking lot in Hong Kong was 60 ppb, and the level was about the same for NO. Colbeck (1998) reported that concentrations in two parking lots in Colchester, UK were similar to those measured at the curb

side. Exposure of car parking lot users to NO_2 is comparable to that arising in the vicinity of roads with moderate traffic density (~9000 vehicles per day). NO_2 concentrations in one parking lot ranged from 30.4 to 47.1 ppb, while those in the payment booth ranged from 22.5 to 31.4 ppb. Rundell et al. (2006) reported PM_1, NO_2, SO_2, CO, and O_3 concentrations at four elementary school playgrounds and one university soccer field in Pennsylvania. NO_2 concentrations were below 100 ppb. The number concentration in the PM_1 size fraction decreased with distance away from the highway (from 140,000 number/cm^3 within 10 m of the road to 40,000 number/cm^3 at 80 m).

Indoor environments, which are close to traffic, include buildings and houses along major, busy roads. Most studies show that indoor NO_2 is correlated with outdoor NO_2, and is also a function of distance to traffic, traffic density and meteorological parameters. The level of indoor NO_2 in those microenvironments is also affected by indoor sources. Bae et al. (2004) reported indoor and outdoor concentrations of NO_2 in 32 shoe stalls in Seoul, which were located on busy streets. Working-hour (10 ± 2.1 h) NO_2 was measured by Yanagisawa passive filter badges. Mean indoor and outdoor NO_2 concentrations were 57.4 and 58.1 ppb with a mean indoor vs. outdoor ratio of 0.93. Maximum indoor and outdoor NO_2 concentrations were 94.1 and 96.3 ppb. In this study, outdoor traffic generated NO_2 is likely the main source of indoor exposures due to the lack of indoor NO_2 sources. Outdoor and in-classroom NO_2 were measured using Palmes tubes during three 2-week periods in six city districts near motor ways in the West of the Netherlands (Roorda-Knape et al., 1998). NO_2 concentrations in classrooms were significantly correlated with car and total traffic density ($r_p = 0.68$), percentage of time downwind ($r_p = 0.88$) and distance of the school from the motorway ($r_p = -0.83$). Cotterill et al. (1997) measured indoor and outdoor NO_2 in 40 homes in Huddersfield, UK, over three consecutive two-week periods in late 1994 using Palmes tubes. The authors found that proximity to a main road had little effect on indoor levels of nitrogen dioxide (a mean of 1 ppb indoor concentration difference was found for homes close to main roads and homes close to side roads). A t-test suggested that there was no difference in indoor levels of nitrogen dioxide due to proximity to the main road after indoor sources were controlled by the type of cookers. In this study, meteorological parameters were measured, but meteorological parameters were not controlled during data analysis.

Personal exposure is determined by both indoor and outdoor levels of NO_2. Most studies show significant associations between personal exposure and the traffic density. The influence of indoor sources on personal exposure was also observed in those studies. Alm et al. (1998) reported the weekly personal NO_2 exposures of 246 children aged 3-6 years in Helsinki. Weekly personal exposures were measured for 13 weeks in winter and spring in 1991 using Palmes tubes. The 13 week geometric mean of the NO_2 exposures was higher for the children living in the downtown (13.9 ppb) than in the suburban area (9.2 ppb, $p = 0.0001$). Rotko et al. (2001) reported the EXPOLIS-Helsinki study results and observed that the NO_2 exposure was significantly associated with traffic volume near homes. The average exposure level of 138 subjects having low or moderate traffic near their homes was 12.3 ppb, while the level was 15.8 ppb for the 38 subjects having high traffic volume near home. Gauvin et al. (2001) reported the VESTA study results. An index of traffic density and proximity was constructed as the ratio of traffic density to distance from a roadway. The index was one of the significant interpreters of personal exposure in all three cities ($p < 0.05$ for Grenoble and Toulouse, and $0.05 < p < 0.15$ for Paris). Kodama et al. (2002) showed that personal exposure was similar to residential home NO_2 concentration for residences along busy roads. The authors also observed that personal exposure levels were higher than outdoor levels during the winter, while during the summer, personal exposure levels were lower than ambient levels, due to the influence of indoor sources and low ventilations in the winter. Although the personal to outdoor relationship was dominated by indoor sources, the effects of outdoor NO_2 on personal exposure could still be observed after controlling the indoor source effects. Nakai et al. (1995) observed that personal exposure levels basically followed the ambient concentrations patterns given above; i.e., exposures in Zone A (0-20 m from the road) were the highest and exposures in Zone C (the suburban background area) were the lowest for residents not using an unvented heater (as defined before, Zone A was 0-20 m from the road; Zone B was 20-150 m from the road. The maximum difference of personal exposure between Zone A and Zone C was approximately 20 ppb. The NO_2 exposure for a special population, athletes, was addressed by Carlisle et al. (2001). The authors pointed out that athletes could be a potential population at risk, if the ambient NO_2 concentration is high because (1) inhalation rate increases during exercise, (2) a large fraction of air is inhaled through the mouth during exercise, effectively bypassing the normal nasal mechanisms for the filtration of large particles and soluble vapors, and (3) the increased air flow velocity carries pollutants

deeper into the respiratory tract and pulmonary diffusion capacity increases during exercise. This might also be true for outdoor workers but few data are available to perform the exposure assessment.

Although traffic is a major source of ambient NO_2, industrial point sources are also contributors to ambient NO_2. However, no published reports were found to address the effect of those sources on population exposure within the United States. Nerriere et al. (2005) measured personal exposures to $PM_{2.5}$, PM_{10}, and NO_2 in traffic dominated, urban background and industrial settings in Paris, Grenoble, Rouen, and Strasbourg, France. They always found highest ambient concentrations and personal exposures close to traffic. In some cases, urban and background, concentrations of NO_2 were higher than in the industrial zone. However, PM levels and personal exposures tended to be higher in the industrial area than in the traffic dominated area. It should be remembered that there can be high traffic emissions in industrial zones, such as in the Ship Channel in Houston, TX. In rural areas where traffic is sparse, other sources could dominate. For example, Martin et al. (2003) found pulses of NO_2 release from agricultural areas following rainfall and there are contributions from wildfires and residential wood burning.

Exposure Reconstruction

Personal exposure has been evaluated in each major microenvironment, where either the NO_2 concentration is high or people spend most of their daily time. As shown in Equation AX3-13, personal exposure could be reasonably reconstructed if we know the NO_2 concentration in each microenvironment and the duration of personal exposure in each microenvironment. Levy et al., (1998a) reconstructed personal exposures measured in an international study with a time-weighted average exposure model (Equation AX3-12). The personal exposure was reconstructed based on the measured NO_2 concentrations in residential indoor, residential outdoor, and workplace microenvironments, and the time people spent in those environments. The mean measured personal NO_2 exposure was 28.8 ppb and a mean of estimated NO_2 exposure was 27.2 ppb. The Spearman correlation coefficient between personal measured exposure and reconstructed exposure was 0.81. The same approach was applied by Kousa et al. (2001) to reconstruct the personal exposures in the EXPOLIS study. A correlation coefficient of 0.86 was observed for the association between measured NO_2 exposure and reconstructed NO_2 exposure (data were log-transformed), and the slope and the intercept were 0.90 and 0.22 respectively for the reconstructed exposure vs. measured exposure. In the two studies mentioned above, NO_2

exposure during commuting was not measured. Probably that is part of the reason why reconstructed NO_2 exposure was lower than the measured NO_2 exposure.

AX3.5.3 Exposure Indicators

Physically, personal exposure levels are determined by those physical parameters in Equations AX3-12 to AX3-16, i.e., the time people spend in each microenvironment and the NO_2 concentrations in each microenvironment, which is determined by source emission strength, air exchange rate, penetration coefficient, the NO_2 decay rate and the volume of the microenvironment. Any factors that can influence the above physical parameters can modify the level of personal exposure. These factors are defined as exposure indicators in this section. The indoor, outdoor and personal NO_2 levels on each stratum of those factors will be summarized.

Those factors can be classified in to the following categories: (1) factors associated with environmental conditions, such as weather and season; (2) factors associated with dwelling conditions, such as the location of the house and ventilation system; (3) factors associated with indoor sources, such as the type of range and the fuel type; (4) factors associated with personal activities, such as the time spent on cooking or commuting; (5) socioeconomic status, such as the level of education and the income level; and (6) demographic factors, such as age and gender.

Most studies addressed the influences of dwelling condition and indoor sources on indoor and personal exposures. A few studies explored the impacts of environmental factors and personal activities on personal exposures. Indoor and personal exposures have rarely been stratified by socioeconomic and demographic factors. Indoor, outdoor, and personal exposure levels are presented in Table AX3.18, stratified by environmental factors, dwelling conditions, indoor sources, and personal activities factors. The effects of socioeconomic and demographic factors on the indoor, outdoor, and personal levels are summarized in Table AX3.19.

Season is an environmental factor affecting both indoor and outdoor levels, and thus personal NO_2 levels. During the wintertime, the mixing height is usually lower than during the summer, and therefore concentrations of many primary pollutants are higher than in the summer. Wintertime is also a heating season, which usually leads to higher indoor source emissions and lower air exchange rates. Therefore, a higher indoor NO_2 concentration can be expected during the winter. For most cases, the differences of indoor or personal NO_2 exposure between the heating and non-heating season are within several ppb, but sometimes the difference could be

close to 20 ppb (Zota et al., 2005). Other environmental factors include day of the week (weekday versus weekend), and the wind direction, as shown in Table AX3.18.

The dwelling conditions are also associated with indoor, outdoor, and personal NO_2 levels. Location of the dwelling unit is an indicator of ambient NO_2 source strength. A house located in an urban center or close to a major road is expected to have higher outdoor and indoor NO_2 levels, and the differences in NO_2 exposures are often within 20 ppb based on passive sampler monitoring. The age of the house, house type, and window type can affect the ventilation of dwelling units, and sometimes the type of heating and cooking applicances in a house. Range and fuel type are the indoor source factors discussed the most in the literature. It is common to see differences larger than 10 ppb in indoor and personal NO_2 exposures between a gas range home (especially gas range with pilot light) and an electric range home. Sometimes the differences could be as high as 40 ppb. For peak short-term exposures, the difference could reach 100 ppb.

The level of personal exposure is dependent upon the time a person spends in each microenvironment. Kawamoto et al. (1997), Levy et al. (1998a), and Chao and Law (2000) clearly showed that personal NO_2 exposure increases with time spent cooking or commuting.

The common findings are summarized above. However, there are inconsistencies in the literature. For example, smoking is claimed to be a significant factor in some studies but not in others, and the same can be said for proximity to a major road. For another example, a higher indoor NO_2 level could be found in a rural home rather than in an urban home (Table AX3.18), although most studies found the opposite. Part of the reason is that exposure indicators function together, as a multidimensional parameter space, on indoor and personal exposures. They are not independent of each other. Unfortunately, studies have rarely been conducted to understand the associations between these exposure indicators and to use the study findings to explain indoor and personal NO_2 exposures.

More effort put on exposure indicator studies should help in finding better surrogate measurements for personal exposures. Although misclassifying exposures in epidemiological studies is almost inevitable, and it is unlikely that the personal exposures of all subjects will be measured, a better knowledge of the effects of exposure indicators on personal exposure will help reduce exposure errors in exposure and epidemiological studies and help interpret those study results.

AX3.6 CONFOUNDING AND SURROGATE ISSUES

Confounding is the technical term for finding an association for the wrong reason. It is associated with both the exposure and the disease being studied, but is not a consequence of the exposure. The confounder does not need to be an exposure for the disease under study. The confounding variable can either inflate or deflate the true relative risk.

Since epidemological studies of NO_2 often use ambient concentrations to reflect exposures, whether confounding of NO_2 findings is possible can be determined by examining associations among ambient concentrations and personal exposures to NO_2 and its relevant copollutants. Importantly, by examining these associations, it is also possible to evaluate whether a copollutant may act as a confounder or as a proxy of ambient NO_2 concentrations.

The potential for confounding of ambient NO_2 health effects is discussed in terms of four relationships: (1) ambient NO_2 and ambient copollutant concentrations, (2) personal NO_2 and personal copollutant exposures, (3) personal NO_2 exposures and ambient copollutant concentrations, and (4) ambient NO_2 concentrations and personal copollutant exposures.

1) Associations between Ambient NO_2 and Ambient Copollutant Concentrations

Confounding of NO_2 health effects is often examined at the ambient level, since ambient concentrations are generally used to reflect exposures in epidemiological studies. The majority of studies examining pollutant associations in the ambient environment have focused on ambient NO_2, $PM_{2.5}$ (and its components), and CO, with fewer studies reporting the relationship between ambient NO_2 and ambient O_3 or SO_2.

Correlations between concentrations of ambient NO_2 and other ambient pollutants, $PM_{2.5}$ (and its components where available), CO, O_3 and SO_2 are summarized in Table AX3.20. Data were compiled from Environmental Protection Agency's Air Quality System and a number of exposure studies. Mean values of site-wise correlations are shown. As can be seen from the table, NO_2 is moderately correlated with $PM_{2.5}$ (range: 0.37 to 0.78) and with CO (0.41 to 0.76) in suburban and urban areas. At rural locations, such as Riverside, CA, associations between ambient NO_2 and ambient CO concentrations (both largely traffic-related pollutants) are much lower, likely as the result of other sources of both CO and NO_2 increasing in importance in rural areas. These sources include oxidation of CH_4 and other biogenic compounds, wood burning and wildfires (for CO); and soil emissions, lightning, and wood burning and wildfires for NO_2.

In urban areas, the ambient NO_2-CO correlations vary widely. The strongest correlations are seen between NO_2 and elemental carbon. Note that the results of Hochadel et al. (2006) for $PM_{2.5}$ optical absorbance have been interpreted in terms of elemental carbon (EC). Correlations between ambient NO_2 and ambient O_3 are mainly negative, with again considerable variability in the observed correlations. Only one study (Sarnat et al., 2001) examined associations between ambient NO_2 and ambient SO_2 concentrations, showing a negative correlation during winter. The robustness of this result needs to be examined in other cities.

Kim et al. (2006) reported the associations between 24 h averaged NO_2 and other pollutions for personal exposures and ambient concentrations in a study in Toronto, Canada from August 1999 to November 2001. The median, mean, and standard deviation of the correlations between ambient NO_2 and ambient $PM_{2.5}$ were 0.52, 0.44, and 0.35 respectively; and 0.81, 0.72, and 0.22 respectively for the correlation between NO_2 and CO.

In an exposure study in Steubenville, Ohio, Sarnat et al. (2006) reported the associations between ambient concentrations and personal exposures for different pollutants. Ambient NO_2 was significantly associated with ambient $PM_{2.5}$, sulfate and EC during the fall (slope = 0.38, 0.96, and 7.01; and R^2 = 0.61, 0.49, and 0.68 respectively) but not during the summer (slope = −0.01, −0.17, and 3.76; and R^2 = 0.0, 0.01, and 0.06 respectively).

In a related study, Connell et al. (2005) reported the correlation between ambient NO_x and $PM_{2.5}$ during a comprehensive air monitoring program in Steubenville, Ohio. Across the two year study (August 2000~April 2002), the Spearman correlation coefficient (r_s) between hourly ambient $PM_{2.5}$ and NO concentrations was 0.33, and between hourly ambient $PM_{2.5}$ and NO_2 concentrations was 0.50. The authors suggested the importance of a common factor influencing ambient concentrations of these species.

Kim et al. (2005) analyzed particle composition and gas phase data collected during the RAPS/RAMS study on St. Louis, MO from 1975 to 1977 in terms of source contributions to $PM_{2.5}$. This study examined the spatial variability of source contributions to $PM_{2.5}$ at the ten monitoring sites in that study.

Sarnat et al. (2001) and reported associations between personal exposures and ambient concentration across pollutants in a study conducted in the Baltimore area. At the ambient level, NO_2 was significantly correlated with $PM_{2.5}$ (r_s = 0.37) and CO (r_s = 0.75) during the summer and with CO (r_s = 0.76), SO_2 (r_s = −0.17), $PM_{2.5}$ (r_s = 0.75) and O_3 (r_s = −0.71) during the winter.

Linn et al. (1996) reported short-term air pollution exposures in Los Angeles area school children. Correlations between different pollutants were weaker: $r_p = 0.11$ for ambient NO_2 and O_3; $r_p = 0.25$ for ambient NO_2 and outdoor $PM_{2.5}$.

Lee et al. (2002) found that ambient NO_2 was significantly correlated with O_3 ($r_p = -0.34$).

Foreign Studies

Hochadel et al. (2006) reported the results of research which is part of a cohort study on the impact of traffic-related air pollution on respiratory health, conducted at the western end of the Ruhr-area in North-Rhine Westphalia, Germany. Strong correlations across the measurement sites were observed between annual average $PM_{2.5}$ absorbance and NO_2 concentrations ($r_p = 0.93$), whereas $PM_{2.5}$ mass concentration was less strongly correlated with NO_2 ($r_p = 0.41$). The only major absorbing agent in $PM_{2.5}$ is elemental carbon (EC) as other components (sulfate, nitrate, organic carbon) either do not absorb or at best are only weakly absorbing. Therefore, correlations between $PM_{2.5}$ absorbance and NO_2 may be inferred as correlations between EC and NO_2.

Hazenkamp-von Ark et al. (2004) reported the $PM_{2.5}$ and NO_2 associations across 21 European study centers during ECRHS II. The correlation between annual NO_2 and $PM_{2.5}$ concentrations is fair (Spearman correlation coefficient $r_s = 0.75$), but when considered as monthly means, the correlation is far less consistent and varies substantially between centers. The authors pointed out that NO_2 is attributed to traffic emissions, a relatively constant source of pollution throughout the year. $PM_{2.5}$ on the other hand, can be driven by other sources such as wind-blown dust, although usually it consists predominantly of primary and secondary particles from combustion processes. Sources, such as Saharan dust in Spain, probably cause some of the observed patterns. The wide range of correlations between $PM_{2.5}$ and NO_2 evokes the hypothesis that monthly $PM_{2.5}$ mass concentrations in some centers may be driven by traffic emissions, whereas in other centers, particles from other sources may be of further relevance.

Cyrys et al. (2003) reported the results of a source apportionment study in Erfurt, Germany. Hourly NO_2 was correlated with NO, CO, $PM_{0.01\text{-}2.5}$ number concentration, and $PM_{0.01\text{-}2.5}$ mass concentration ($r_p = 0.73$, 0.74, 0.55, and 0.50 respectively). Stronger correlations were found daily correlation between NO_2 and NO, CO, $PM_{0.01\text{-}2.5}$ number concentration, and

$PM_{0.01-2.5}$ mass concentration (r_p = 0.87, 0.76, 0.71, and 0.66 respectively). The observed high correlations between CO, NO, and NO_2 indicate that direct emissions from mobile sources might be the major contributors to the concentrations of these gaseous pollutants.

Rojas-Bracho et al. (2002) conducted a study of children's exposures in Santiago, Chile. During the study, indoor, outdoor, and personal $PM_{2.5}$, PM_{10}, $PM_{10-2.5}$, and NO_2 were measured 24 h averaged for five consecutive days). Outdoor NO_2 was significantly associated with all PM fractions (slope = 1.82 and R^2 = 0.59 for $PM_{2.5}$; slope = 3.12 and R^2 = 0.57 for PM_{10}; and slope = 1.11 and R^2 = 0.32 for $PM_{2.5-10}$).

Modig et al. (2004) investigated whether NO_2 can be used to indicate ambient and personal levels of benzene and 1, 3-butadiene in air. The stationary measurements showed strong relations between 1,3-butadiene, benzene and NO_2 (r_p = 0.70 for NO_2 and benzene; and r = 0.77 for NO_2 and 1,3-butadiene). This study supports NO_2 as a potential indicator for 1,3 butadiene and benzene levels in streets or urban background air.

In summary, ambient NO_2 was moderately correlated with corresponding ambient concentrations of its co- pollutants. Based on associations in the ambient environment, results suggest a possibility of confounding of ambient NO_2 health effects by ambient $PM_{2.5}$ (and its components) and by ambient CO.

2) Associations between Personal (NO_2) and Personal Copollutant Exposures

For this section, measured personal NO_2 exposures are regarded as the "true" personal exposure. The correlation between personal NO_2 exposure and personal exposure to other pollutants are summarized below in Table AX3.21.

In Kim et al. (2006), the median, mean and standard deviation of the correlation between NO_2 and $PM_{2.5}$ personal exposures for eleven subjects were 0.43, 0.41, and 0.28 respectively; and 0.16, 0.12, and 0.42 respectively for the correlation between NO_2 and CO (Kim et al., 2006).

Although Sarnat et al. (2001) found that personal exposures to $PM_{2.5}$ were generally not significantly associated with personal exposures to gases in Baltimore, personal NO_2 was significantly associated with personal $PM_{2.5}$ (slope = 0.18, intercept = 18.65, $p < 0.01$, and n = 213) and personal $PM_{2.5}$ of ambient origin (slope = 0.17, intercept = 12.77, $p < 0.05$, and n = 150) during the summer. There was some evidence to indicate that the strength of the association was driven largely by the cohort of older adult subjects, and not by the children's or COPD patients cohorts. They noted that gas stove usage did not significantly affect personal

NO_2 to $PM_{2.5}$ relations, but did affect relations between personal NO_2 and personal $PM_{2.5}$ of ambient origin. They further pointed out that associations observed among pollutants in ambient air may not be reflected in personal exposures and that they may not persist across seasons. However, Lai et al. (2004) found that personal exposure to NO_2 was slightly negatively correlated with personal exposure to $PM_{2.5}$ and total VOCs in a study conducted from 1998 to 2000 in Oxford, UK (−0.1 for $PM_{2.5}$, 0.3 for CO, and −0.11 for TVOCs).

Modig et al. (2004) investigated whether NO_2 can be used to indicate ambient and personal levels of benzene and 1, 3-butadiene in air. The results from the personal measurements showed a negligible association of NO_2 with 1,3-butadiene ($r_p = 0.06$) as well as with benzene ($r_p = 0.10$), while the correlation coefficient between benzene and 1,3-butadiene was high and significant ($r_p = 0.67$). The weak relations found for the personal measurements do not support the use of NO_2 as an indicator for personal 1,3-butadiene and benzene exposure. Although gas stove and kerosene heaters were almost absent in the study area, this study included both smokers and non-smokers, but the data were not stratified. Smoking is a major source of both benzene and 1,3-butadiene, in addition to motor vehicles. If smoking were the major cause of the poor association between NO_2 and the gases in the personal measurements, then this would indicate that smoking was not a major source of personal NO_2. Thus, this study cannot determine whether personal NO_2 is an indicator of traffic generated VOCs and so the interpretation of results in this paper is problematic.

In the Paris office worker study, no relation was observed between personal NO_2 and $PM_{2.5}$ exposures ($r_p = 0.12$, n = 53, p = 0.38) (Mosqueron et al., 2002). In addition, NO_2 and $PM_{2.5}$ concentrations were correlated neither in-home ($r_p = 0.06$, n = 54, p = 0.69) nor in-office ($r_p = 0.05$, n = 55, p = 0.74).

Associations with HONO

Spicer et al. (1993) and Wainman et al. (2000) suggested the presence of a strong indoor source of HONO from heterogeneous reactions involving NO_2 and water films on indoor surfaces. Hence, combustion appliances are sources for exposures to both NO_2 exposure and HNO_2. Epidemiological studies of NO_2 health effects should consequently consider the potential confounding effects of NO_2 and vice versa.

Jarvis et al. (2005) reported the indoor nitrous acid and lung function in adults as part of European Community Respiratory Health Survey (ECRHS). Indoor HONO and indoor and outdoor NO_2 were measured. Indoor NO_2 were correlated with HONO ($r_p = 0.77$) but no significant association of indoor NO_2 with symptoms or lung function was observed.

Lee et al. (2002) studied the nitrous acid, nitrogen dioxide, and ozone concentrations in residential environments. The authors found that indoor NO_2 was significantly correlated with HONO ($r_p = 0.511$).

As shown above, very few studies showed the relationship between personal NO_2 exposure and other pollutant exposures. In general, personal NO_2 was moderately correlated with $PM_{2.5}$ and CO. Due to the lack of personal HONO exposure data, indoor HONO was used as an indicator for personal exposure, and current studies showed that indoor HONO was correlated with indoor NO_2 with high correlation coefficients, which suggested that the collect ion of HONO exposure data would help interpret adverse health outcome in the NO_2 health risk assessment.

3) Personal (NO_2) -Ambient Copollutants

The relationship between personal NO_2 exposure and other ambient pollutants are summarized in Table AX3.22.

In Steubenville, Ohio, Sarnat et al. (2006) found that personal NO_2 was significantly associated with ambient $PM_{2.5}$ and ambient sulfate during the fall (slope = 0.17 and $R^2 = 0.21$ for $PM_{2.5}$; and slope = 0.34 and $R^2 = 0.12$ for sulfate); and was significantly associated with ambient EC in both summer and fall (slope = 1.81 and $R^2 = 0.03$ for the summer; and slope = 3.71 and $R^2 = 0.32$ for the fall).

Kim et al. (2006) also reported correlations between personal exposure and ambient measurements across pollutants. The median, mean, and standard deviation of the correlation between personal NO_2 and ambient $PM_{2.5}$ were 0.36, 0.30, and 0.30 respectively; and 0.17, 0.20, and 0.41 respectively for the correlation between personal NO_2 and ambient CO. The authors suggested that the existing correlation between $PM_{2.5}$ and NO_2 for both ambient measurements and personal exposures suggests that there is potential for NO_2 to be a confounder of $PM_{2.5}$, and vice versa. Therefore, it may be appropriate for time-series epidemiological studies to control for confounding by NO_2 in $PM_{2.5}$ risk models, and vice versa.

In a study conducted in Santiago, Chile (Rojas-Bracho et al., 2002) personal NO_2 was moderately associated with $PM_{2.5}$ (slope = 1.99 and $r^2 = 0.42$) and PM_{10} (slope = 2.13 and $r^2 = 0.15$) but not coarse particles. At the indoor level, the same observation held (slope = 0.86 and $r^2 = 0.22$ for $PM_{2.5}$; slope = 1.0 and $R^2 = 0.2$ for PM_{10}). "However, in comparing the indoor and outdoor associations, we find that the latter is more highly significant and that the intercept is smaller. It is likely that in outdoor environments, there are more high-temperature combustion processes, which are associated with nitrogen oxide emissions. Since nitrogen oxides are precursors of secondary particles, which partly form $PM_{2.5}$, our results showed a stronger association between these two pollutions outdoors."

Lee et al. (2002) studied nitrous acid, nitrogen dioxide, and ozone concentrations in residential environments. The authors found that indoor NO_2 was significantly correlated with outdoor O_3 ($r_p = -0.220$).

These studies above show moderate correlations between personal NO_2 exposure and ambient $PM_{2.5}$, PM_{10}, EC, sulfate, and CO. Based on our knowledge that, moderate to strong personal-ambient correlations exist for all the other pollutants mentioned above all of those species might serve as confounders for NO_2 exposure (detailed evaluation of the personal vs. ambient relationship for these pollutants are beyond the scope of this document).

4) Ambient NO_2-Personal Copollutant

Correlation between ambient NO_2 and personal exposure to copollutants are summarized in Table AX3.23.

Sarnat et al. (2006) found that ambient NO_2 was significantly associated with personal $PM_{2.5}$ and personal sulfate during the fall (slope = 0.93 and $R^2 = 0.25$ for $PM_{2.5}$; and slope = 0.28 and $R^2 = 0.27$ for sulfate); and was significantly associated with personal EC during both summer and fall (slope = 0.02 and $R^2 = 0.07$ during the summer; and slope = 0.08 and $R^2 = 0.49$ during the fall) in Steubenville, OH. Sarnat et al. (2006) suggested that for most cases, ambient gas concentrations, although not suitable proxies of gas exposures are equally not suitable for particle exposures in time-series health studies. Despite this, numerous epidemological studies have linked 24-h ambient gas concentrations to adverse health impacts, suggesting that the gases may indeed elicit biological responses alone or in combination with other pollutants, or are acting as proxies for shorter-term exposures. The authors pointed out that

for Steubenville in the fall, a season with strong associations between ambient particle and NO_2 concentrations, the separation of particle and NO_2 health effects in daily time-series studies may be difficult, and more precise exposure metrics may be needed. The authors suggested that personal-ambient relationships are greatly dependent on ambient conditions (e.g., season and meteorology) and behavior (e.g., use of windows). However, further factors such as building design will also be extremely important, further exposure assessment work, particularly in different geographic and climatic zones, is needed.

During both summer and winter in Baltimore (Sarnat et al., 2001), ambient NO_2 was significantly associated with personal $PM_{2.5}$ (slope = 0.42, intercept = 12.38, and n = 225 during the summer; and slope = 0.24, intercept = 13.16, and n = 487 during the winter). Also significant relationships held for ambient NO_2 and personal exposures to $PM_{2.5}$ of ambient origin. Ambient NO_2 was also significantly associated with personal EC (slope = 0.05 and p = 0.0001), as an indicator of mobile source pollution. In conclusion, the authors suggested that ambient gases were acting as surrogates for personal $PM_{2.5}$ exposure instead of confounding effects of personal $PM_{2.5}$ exposure.

Vinzents et al. (2005) found that ambient temperature and NO_2 concentrations at one of the street stations were the only significant predictors of ultra fine particle exposure during bicycling in traffic ($R^2 = 0.74$). Kim et al. (2006) also reported correlations between personal exposure and ambient measurements across pollutants. The median, mean, and standard deviation of the correlation between ambient NO_2 and personal $PM_{2.5}$ were 0.24, 0.29, and 0.33 respectively; and 0.26, 0.22, and 0.32 respectively for the correlation between ambient NO_2 and personal CO.

Studies above shows that ambient NO_2 is moderately correlated with personal EC and ultrafine particle exposures, but only weakly to moderately correlated with personal $PM_{2.5}$ mass and sulfate exposures. Since ambient NO_2 concentrations has been shown to be significant proxy for corresponding personal NO_2 exposures, these findings suggest that ambient NO_2 may be acting as a proxy not only for its own exposures but also to exposures to EC and ultrafine particles. As a result, it may not be possible to separate the health effects of from those of other pollutants, especially from the same source.

In the analysis of the confounding effect of exposure, we are limited by the lack of key data: (1) multipollutant exposure studies were rarely conducted and even fewer studies reported

the cross-level (ambient and personal exposure) and cross-pollutant correlations; (2) most studies focus on a several copollutants (PM and its components, CO, O_3, and some VOCs) with little data available for other possibly important copollutants; (3) the impact of indoor and personal sources on the possibility of confounding has not yet been examined; and (4) the impact of measurement uncertainties, which can be large as mentioned in Section AX3.4.1, on confounding needs to be examined. Finally, the analysis shown above in the exposure assessment should be integrated with other analysis in other parts of the risk assessment.

AX3.7 A FRAMEWORK FOR MODELING HUMAN EXPOSURES TO NO_2 AND RELATED PHOTOCHEMICAL AIR POLLUTANTS

AX3.7.1 Introduction: Concepts, Terminology, and Overall Summary

Predictive (or prognostic) exposure modeling studies[1], specifically focusing on NO_2, could not be identified in the literature, though, often, statistical (diagnostic) analyses have been reported using data obtained in various field exposure studies (see Section AX3.5.1). However, existing prognostic modeling systems for the assessment of inhalation exposures can in principle be directly applied to, or adapted for, NO_2 studies; specifically, such systems include APEX, SHEDS, and MENTOR-1A, to be discussed in the following sections. Nevertheless, it should be mentioned that such applications will be constrained by data limitations, such as the degree of ambient concentration characterization (e.g., concentrations at the local level) and quantitative information on indoor sources and sinks.

Predictive models of human exposure to ambient air pollutants such as NO_2 can be classified and differentiated based upon a variety of attributes. For example, exposure models can be classified as:

- models of potential (typically maximum) outdoor exposure versus models of actual exposures (the latter including locally modified microenvironmental exposures, both outdoor and indoor),
- Population Based Exposure Models (PBEM) versus Individual Based Exposure Models (IBEM),

[1] i.e. assessments that start from emissions and demographic information and explicitly consider the physical and chemical processes of environmental and microenvironmental transport and fate, in conjunction with human activities, to estimate inhalation intake and uptake.

- deterministic versus probabilistic (or statistical) exposure models,
- observation-driven versus mechanistic air quality models (see Section AX3.7.3 for discussions about the construction, uses and limitations of this class of mathematical models.

Some points should be made regarding terminology and essential concepts in exposure modeling, before proceeding to the overview of specific developments reported in the current research literature:

First, it must be understood that there is significant variation in the definitions of many of the terms used in the exposure modeling literature; indeed, the science of exposure modeling is a rapidly evolving field and the development of a standard and commonly accepted terminology is an ongoing process (see, e.g., WHO, 2004).

Second, it should also be mentioned that, very often, procedures that are called exposure modeling, exposure estimation, etc. in the scientific literature, may in fact refer to only a sub-set of the complete set of steps or components required for a comprehensive exposure assessment. For example, certain self-identified exposure modeling studies focus solely on refining the sub-regional or local spatio-temporal dynamics of pollutant concentrations (starting from raw data representing monitor observations or regional grid-based model estimates). Though not exposure studies per se, such efforts have value and are included in the discussion of the next sub-section, as they provide potentially useful tools that can be used in a complete exposure assessment. On the other hand, formulations that are self-identified as exposure models but actually focus only on ambient air quality predictions, such as chemistry-transport models, are not included in the discussion that follows.

Third, the process of modeling human exposures to photochemical pollutants (traditionally focused on ozone) is very often identified explicitly with population-based modeling, while models describing the specific mechanisms affecting the exposure of an actual individual (at specific locations) to an air contaminant (or to a group of co-occurring gas and/or aerosol phase pollutants) are usually associated with studies focusing specifically on indoor air chemistry modeling.

Finally, fourth, the concept of microenvironments, introduced in earlier sections of this document, should be clarified further, as it is critical in developing procedures for exposure modeling. In the past, microenvironments have typically been defined as individual or aggregate

locations (and sometimes even as activities taking place within a location) where a homogeneous concentration of the pollutant is encountered. Thus a microenvironment has often been identified with an ideal (i.e. perfectly mixed) compartment of classical compartmental modeling. More recent and general definitions view the microenvironment as a control volume, either indoors or outdoors, that can be fully characterized by a set of either mechanistic or phenomenological governing equations, when appropriate parameters are available, given necessary initial and boundary conditions. The boundary conditions typically would reflect interactions with ambient air and with other microenvironments. The parameterizations of the governing equations generally include the information on attributes of sources and sinks within each microenvironment. This type of general definition allows for the concentration within a microenvironment to be non-homogeneous (non-uniform), provided its spatial profile and mixing properties can be fully predicted or characterized. By adopting this definition, the number of microenvironments used in a study is kept manageable, but variability in concentrations in each of the microenvironments can still be taken into account. Microenvironments typically used to determine exposure include indoor residential microenvironments, other indoor locations (typically occupational microenvironments), outdoors near roadways, other outdoor locations, and in-vehicles. Outdoor locations near roadways are segregated from other outdoor locations (and can be further classified into street canyons, vicinities of intersections, etc.) because emissions from automobiles alter local concentrations significantly compared to background outdoor levels. Indoor residential microenvironments (kitchen, bedroom, living room, etc. or aggregate home microenvironment) are typically separated from other indoor locations because of the time spent there and potential differences between the residential environment and the work/public environment.

Once the actual individual and relevant activities and locations (for Individual Based Modeling), or the sample population and associated spatial (geographical) domain (for Population Based Modeling) have been defined along with the temporal framework of the analysis (time period and resolution), the comprehensive modeling of individual/population exposure to NO_2 (and related pollutants) will in general require seven steps (or components, as some of them do not have to be performed in sequence) that are listed below. This list represents a composite based on approaches and frameworks described in the literature over the last twenty-five years (Ott, 1982; Ott, 1985; Lioy, 1990; U.S. Environmental Protection Agency, 1992;

Georgopoulos and Lioy, 1994; U.S. Environmental Protection Agency, 1997; Buck et al., 2003; Price et al., 2003; Georgopoulos et al., 2005; WHO, 2005; U.S. Environmental Protection Agency, 2006a; Georgopoulos and Lioy, 2006) as well on the structure of various inhalation exposure models (NEM/pNEM, HAPEM, SHEDS, REHEX, EDMAS, MENTOR, ORAMUS, APEX, AIRPEX, AIRQUIS, etc., to be discussed in the following section) that have been used in the past or in current studies to specifically assess inhalation exposures. Figure AX3.23, adapted from Georgopoulos et al. (2005), schematically depicts the sequence of steps involved that are summarized here (and further discussed in the following sub-sections).

1. Estimation of the background or ambient levels of both NO_2 and related photochemical pollutants. This is done through either (or a combination of):

    a. multivariate spatio-temporal analysis of fixed monitor data, or

    b. emissions-based, photochemical, air quality modeling (typically with a regional, grid-based model such as Models-3/CMAQ or CAMx) applied in a coarse resolution mode.

2. Estimation of local outdoor pollutant levels of both NO_2 and related photochemical pollutants. These levels could typically characterize the ambient air of either an administrative unit (such as a census tract, a municipality, a county, etc.) or a conveniently defined grid cell of an urban scale air quality model. Again, this may involve either (or a combination of):

    a. spatio-temporal statistical analysis of monitor data, or

    b. application of an urban multi-scale, grid based model (such as CMAQ or CAMx) at its highest resolution (typically around 2-4 km), or

    c. correction of the estimates of the regional model using some scheme that adjusts for observations and/or for subgrid chemistry and mixing processes.

3. Characterization of relevant attributes of the individuals or populations under study (residence and work locations, occupation, housing data, income, education, age, gender, race, weight, and other physiological characteristics). For Population Based Exposure Modeling (PBEM) one can either:

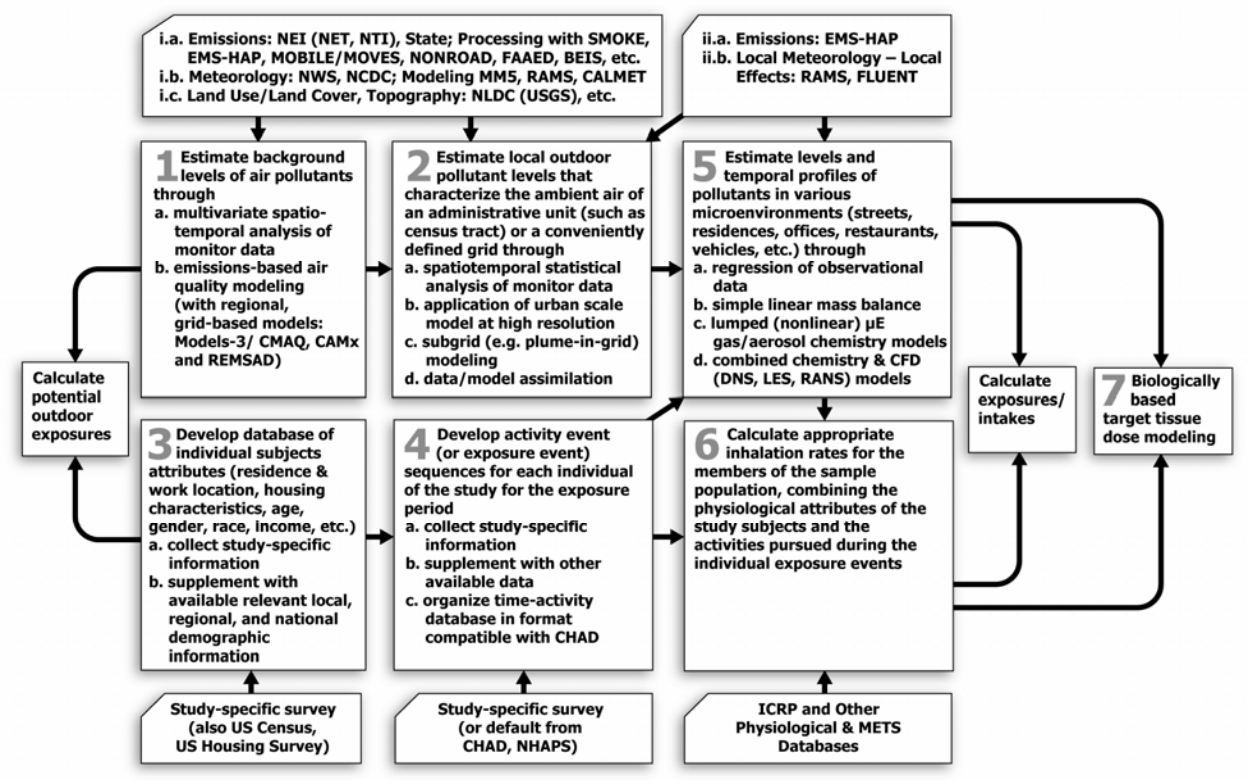

Figure AX3.23. Schematic description of a general framework identifying the processes (steps or components) involved in assessing inhalation exposures and doses for individuals and populations. In general terms, existing comprehensive exposure modeling systems such as SHEDS, APEX, and MENTOR-1A follow this framework.

Source: Figure adapted with modifications from Georgopoulos et al. (2005).

a. select a fixed-size sample population of virtual individuals in a way that statistically reproduces essential demographics (age, gender, race, occupation, income, education) of the administrative population unit used in the assessment (e.g., a sample of 500 people is typically used to represent the demographics of a given census tract, whereas a sample of about 10,000 may be needed to represent the demographics of a county), or

b. divide the population-of interest into a set of cohorts representing selected subpopulations where the cohort is defined by characteristics known to influence exposure.

4. Development of activity event (or exposure event) sequences for each member of the sample population (actual or virtual) or for each cohort for the exposure period. This could utilize:

    a. study-specific information, if available

    b. existing databases based on composites of questionnaire information from past studies

    c. time-activity databases, typically in a format compatible with U.S. Environmental Protection Agency's Consolidated Human Activity Database (CHAD - McCurdy et al., 2000)

5. Estimation of levels and temporal profiles of both NO_2 and related photochemical pollutants in various outdoor and indoor microenvironments such as street canyons, roadway intersections, parks, residences, offices, restaurants, vehicles, etc. This is done through either:

    a. linear regression of available observational data sets,

    b. simple mass balance models (with linear transformation and sinks) over the volume (or a portion of the volume) of the microenvironment,

    c. lumped (nonlinear) gas or gas/aerosol chemistry models, or

    d. detailed combined chemistry and Computational Fluid Dynamics modeling.

6. Calculation of appropriate inhalation rates for the members of the sample population, combining the physiological attributes of the (actual or virtual) study subjects and the activities pursued during the individual exposure events.

7. Calculation of target tissue dose through biologically based modeling estimation (specifically, respiratory dosimetry modeling in the case of NO_2 and related reactive photochemical pollutants) if sufficient information is available.

Implementation of the above framework for comprehensive exposure modeling has benefited significantly from recent advances and expanded availability of computational technologies such as Relational Database Management Systems (RDBMS) and Geographic Information Systems (GIS) (Purushothaman and Georgopoulos, 1997, 1999a,b; Georgopoulos et al., 2005).

In fact, only relatively recently comprehensive, predictive, inhalation exposure modeling studies for ozone, PM, and various air toxics, have attempted to address/incorporate all the components of the general framework described here. In practice, the majority of past exposure modeling studies have either incorporated only subsets of these components or treated some of them in a simplified manner, often focusing on the importance of specific factors affecting exposure. Of course, depending on the objective of a particular modeling study, implementation of only a limited number of steps may be necessary. For example, in a regulatory setting, when comparing the relative effectiveness of emission control strategies, the focus can be on expected changes in ambient levels (corresponding to those observed at NAAQS monitors) in relation to the density of nearby populations. The outdoor levels of pollutants, in conjunction with basic demographic information, can thus be used to calculate upper bounds of population exposures associated with ambient air (as opposed to total exposures that would include contributions from indoor sources) useful in comparing alternative control strategies. Though the metrics derived would not be quantitative indicators of actual human exposures, they can serve as surrogates of population exposures associated with outdoor air, and thus aid in regulatory decision making concerning pollutant standards and in studying the efficacy of emission control strategies. This approach has been used in studies performing comparative evaluations of regional and local emissions reduction strategies in the Eastern U.S (e.g., Purushothaman and Georgopoulos, 1997; Georgopoulos et al., 1997a; Foley et al., 2003).

AX3.7.2 Population Exposure Models: Their Evolution and Current Status

Existing comprehensive inhalation exposure models consider the trajectories of individual human subjects (actual or virtual), or of appropriately defined cohorts, in space and time as sequences of exposure events. In these sequences each event is defined by time, a geographic location, a microenvironment, and the activity of the subject. US Environmental Protection Agency offices (OAQPS and NERL) have supported the most comprehensive efforts in developing models implementing this general concept (see, e.g., Johnson, 2002), and these efforts have resulted in the NEM/pNEM (National Exposure Model and Probabilistic National Exposure Model - Whitfield et al., 1997), HAPEM (Hazardous Air Pollutant Exposure Model - Rosenbaum, 2005), SHEDS (Simulation of Human Exposure and Dose System - Burke et al., 2001), APEX (Air Pollutants Exposure model – US Environmental Protection Agency, 2006b,c),

and MENTOR (Modeling Environment for Total Risk studies - Georgopoulos et al., 2005; Georgopoulos and Lioy, 2006) families of models. European efforts have produced some formulations with similar general attributes as the above U.S. models but, generally, involving simplifications in some of their components. Examples of European models addressing exposures to photochemical oxidants (specifically ozone) include the AirPEx (Air Pollution Exposure) model (Freijer et al., 1998), which basically replicates the pNEM approach and has been applied to the Netherlands, and the AirQUIS (Air Quality Information System) model (Clench-Aas et al., 1999).

The NEM/pNEM, SHEDS, APEX, and MENTOR-1A (MENTOR for One-Atmosphere studies) families of models provide exposure estimates defined by concentration and breathing rate for each individual exposure event, and then average these estimates over periods typically ranging from one h to one year. These models allow simulation of certain aspects of the variability and uncertainty in the principal factors affecting exposure. An alternative approach is taken by the HAPEM family of models that typically provide annual average exposure estimates based on the quantity of time spent per year in each combination of geographic locations and microenvironments. The NEM, SHEDS, APEX, and MENTOR-type models are therefore expected to be more appropriate for pollutants with complex chemistry such as NO_2, and could provide useful information for enhancing related health assessments.

More specifically, regarding the consideration of population demographics and activity patterns:

1. pNEM divides the population of interest into representative cohorts based on the combinations of demographic characteristics (age, gender, and employment), home/work district, residential cooking fuel and replicate number, and then assigns activity diary record from CHAD (Consolidated Human Activities Database) to each cohort according to demographic characteristic, season, day-type (weekday/weekend) and temperature.
2. HAPEM6 divides the population of interest into demographic groups based on age, gender and race, and then for each demographic group/day-type (weekday/weekend) combination, select multiple activity patterns randomly (with replacement) from CHAD and combine them to find the averaged annual time allocations for group members in each census tract for different day types.
3. SHEDS, APEX, and MENTOR-1A generate population demographic files, which contain a user-defined number of person records for each census tract of the population based on proportions of characteristic variables (age, gender, employment, and housing) obtained from the population of interest, and then assign a matching activity diary record from CHAD to each individual record of

the population based on the characteristic variables. It should be mentioned that, in the formulations of these models, workers may commute from one census tract to another census tract for work. So, with the specification of commuting patterns, the variation of exposure concentrations due to commuting between different census tracts can be captured.

The essential attributes of the pNEM, HAPEM, APEX, SHEDS, and MENTOR-1A models are summarized in Table AX3.24.

The conceptual approach originated by the SHEDS models was modified and expanded for use in the development of MENTOR-1A (Modeling Environment for Total Risk – One Atmosphere). Flexibility was incorporated into this modeling system, such as the option of including detailed indoor chemistry of the O_3-NO_x system and other relevant microenvironmental processes, and providing interactive linking with CHAD for consistent definition of population characteristics and activity events (Georgopoulos et al., 2005).

NEM/pNEM implementations have been extensively applied to ozone studies in the 1980s and 1990s. The historical evolution of the pNEM family of models of OAQPS started with the introduction of the first NEM model in the 1980's (Biller et al., 1981). The first such implementations of pNEM/O_3 in the 1980's used a regression-based relationship to estimate indoor ozone concentrations from outdoor concentrations. The second generation of pNEM/O_3 was developed in 1992 and included a simple mass balance model to estimate indoor ozone concentrations. A report by Johnson et al. (2000) describes this version of pNEM/O_3 and summarizes the results of an initial application of the model to 10 cities. Subsequent enhancements to pNEM/O_3 and its input databases included revisions to the methods used to estimate equivalent ventilation rates, to determine commuting patterns, and to adjust ambient ozone levels to simulate attainment of proposed NAAQS. During the mid-1990's, Environmental Protection Agency applied updated versions of pNEM/O_3 to three different population groups in selected cities: (1) the general population of urban residents, (2) outdoor workers, and (3) children who tend to spend more time outdoors than the average child. This version of pNEM/O_3 used a revised probabilistic mass balance model to determine ozone concentrations over one-h periods in indoor and in-vehicle microenvironments (Johnson, 2001).

In recent years, pNEM has been replaced by (or "evolved to") the Air Pollution Exposure Model (APEX). APEX differs from earlier pNEM models in that the probabilistic features of the model are incorporated into a Monte Carlo framework (Langstaff, 2007; US Environmental

Protection Agency, 2006b,c). Like SHEDS and MENTOR-1A, instead of dividing the population-of-interest into a set of cohorts, APEX generates individuals as if they were being randomly sampled from the population. APEX provides each generated individual with a demographic profile that specifies values for all parameters required by the model. The values are selected from distributions and databases that are specific to the age, gender, and other specifications stated in the demographic profile. Environmental Protection Agency has applied APEX to the study of exposures to ozone and other criteria pollutants; APEX can be modified and used for the estimation of NO_2 exposures, if required.

Reconfiguration of APEX for use with NO_2 or other pollutants would require significant literature review, data analysis, and modeling efforts. Necessary steps include determining spatial scope and resolution of the model; generating input files for activity data, air quality and temperature data; and developing definitions for microenvironments and pollutant-microenvironment modeling parameters (penetration and proximity factors, indoor source emissions rates, decay rates, etc.) (ICF Consulting 2005, Decision Points for Configuring APEX for Air Toxics Exposure Assessments). To take full advantage of the probablistic capabilities of APEX, distributions of model input parameters should be used wherever possible.

AX3.7.3 Characterization of Ambient Concentrations of NO_2 and Related Air Pollutants

As mentioned earlier, background and regional outdoor concentrations of pollutants, over a study domain, may be estimated either through emissions-based mechanistic modeling, through ambient-data-based modeling, or through a combination of both. Emissions-based models calculate the spatio-temporal fields of the pollutant concentrations using precursor emissions and meteorological conditions as inputs. The ambient-data-based models typically calculate spatial or spatio-temporal distributions of the pollutant through the use of interpolation schemes, based on either deterministic or stochastic models for allocating monitor station observations to the nodes of a virtual regular grid covering the region of interest. The geostatistical technique of kriging provides various standard procedures for generating an interpolated spatial distribution for a given time, from data at a set of discrete points. Kriging approaches were evaluated by Georgopoulos et al. (Georgopoulos et al., 1997b) in relation to the calculation of local ambient ozone concentrations for exposure assessment purposes, using either monitor observations or regional/urban photochemical model outputs. It was found that kriging is severely limited by the

nonstationary character of the concentration patterns of reactive pollutants; so the advantages this method has in other fields of geophysics do not apply here. The above study showed that the appropriate semivariograms had to be hour-specific, complicating the automated reapplication of any purely spatial interpolation over an extended time period.

Spatio-temporal distributions of pollutant concentrations, such as ozone, PM, and various air toxics have alternatively been obtained using methods of the Spatio-Temporal Random Field (STRF) theory (Christakos and Vyas, 1998a,b). The STRF approach interpolates monitor data in both space and time simultaneously. This method can thus analyze information on temporal trends, which cannot be incorporated directly in purely spatial interpolation methods such as standard kriging. Furthermore, the STRF method can optimize the use of data that are not uniformly sampled in either space or time. STRF was further extended within the Bayesian Maximum Entropy (BME) framework and applied to ozone interpolation studies (Christakos and Hristopulos, 1998; Christakos and Kolovos, 1999; Christakos, 2000). It should be noted that these studies formulate an over-arching scheme for linking air quality with population dose and health effects; however they are limited by the fact that they do not include any microenvironmental effects. MENTOR has incorporated STRF/BME methods as one of the steps for performing a comprehensive analysis of exposure to ozone and PM (Georgopoulos et al., 2005).

Subgrid spatial variability is a major issue with respect to characterizing local concentrations of NO_2. Indeed, the fast rates of the reactions involving the O_3-NO_x system result in significant concentration gradients in the vicinity of sources of NO_x. These gradients are not resolved directly by currently operational grid photochemical air quality simulation models (PAQSMs) such as CMAQ and CAMx. However, both these models include a plume-in-grid. (PinG) option (AER, 2004; Emery and Yarwood, 2005; Gillani and Godowitch, 1999; US Environmental Protection Agency, 2006d) that can be used for large point NO_x sources (such as smokestacks). Nevertheless, PinG formulations typically will resolve gradients in upper atmospheric layers and thus are not necessarily relevant to human exposure calculations, which are affected by gradients caused by a multiplicity of smaller ground level or near ground level combustion sources such as motor vehicles.

Currently PAQSMs are typically applied with horizontal resolutions of 36 km, 12 km, and 4 km and a surface layer thickness that is typically of the order of 30 m. Though

computationally it is possible to increase the resolution of these simulations, there are critical limits that reflect assumptions inherent in the governing equations for both (a) the fluid mechanical processes embodied in the meteorological models (e.g., typically MM5 and RAMS) that provide the inputs for the PAQSMs, and (b) the dispersion processes which become more complex at fine scales (see, e.g., Georgopoulos and Seinfeld, 1989) and thus cannot be described by simple formulations (such as constant dispersion coefficients) when the horizontal resolutions is 2 km or finer.

Application of PAQSMs to urban domains is further complicated by urban topography, the urban heat island, etc. It is beyond the scope, however, of the present discussion, to overview the various issues relevant to urban fluid dynamics and related transport/fate processes of contaminants. However, the issue of modeling subgrid atmospheric dispersion phenomena within complex urban areas in a consistent manner is a very active research area. Reviews of relevant issues and of available approaches for modeling urban fluid mechanics and dispersion can be found in, e.g., Fernando et al. (2001) and Britter and Hanna (2003).

The issue of subgrid variability (SGV) from the perspective of interpreting and evaluating the outcomes of grid-based, multiscale, PAQSMs is discussed in Ching et al. (2006), who suggest a framework that can provide for qualitative judgments on model performance based on comparing observations to the grid predictions and its SGV distribution. From the perspective of Population Exposure Modeling, the most feasible/practical approach for treating subgrid variability of local concentrations is probably through 1) the identification and proper characterization of an adequate number of outdoor microenvironments (potentially related to different types of land use within the urban area as well as to proximity to different types of roadways) and 2) then, concentrations in these microenvironments will have to be adjusted from the corresponding local background ambient concentrations through either regression of empirical data or various types of local atmospheric dispersion/transformation models. This is discussed further in the next subsection.

AX3.7.4 Characterization of Microenvironmental Concentrations

Once the background and local ambient spatio-temporal concentration patterns have been derived, microenvironments that can represent either outdoor or indoor settings when individuals come in contact with the contaminant of concern (e.g., NO_2) must be characterized. This process

can involve modeling of various local sources and sinks, and interrelationships between ambient and microenvironmental concentration levels. Three general approaches have been used in the past to model microenvironmental concentrations:

- Empirical (typically linear regression) fitting of data from studies relating ambient/local and microenvironmental concentration levels to develop analytical relationships.
- Parameterized mass balance modeling over, or within, the volume of the microenvironment. This type of modeling has ranged from very simple formulations, i.e. from models assuming ideal (homogeneous) mixing within the microenvironment (or specified portions of it) and only linear physicochemical transformations (including sources and sinks), to models incorporating analytical solutions of idealized dispersion formulations (such as Gaussian plumes), to models that take into account aspects of complex multiphase chemical and physical interactions and nonidealities in mixing.
- Detailed Computational Fluid Dynamics (CFD) modeling of the outdoor or indoor microenvironment, employing either a Direct Numerical Simulation (DNS) approach, a Reynolds Averaged Numerical Simulation (RANS) approach, or a Large Eddy Simulation (LES) approach, the latter typically for outdoor situations (see, e.g., Milner et al., 2005; Chang and Meroney, 2003; Chang, 2006).

Parameterized mass balance modeling is the approach currently preferred for exposure modeling for populations. As discussed earlier, the simplest microenvironmental setting corresponds to a homogeneously mixed compartment, in contact with possibly both outdoor/local environments as well as other microenvironments. The air quality of this idealized microenvironment is affected mainly by the following processes:

a. Transport processes: These can include advection/convection and dispersion that are affected by local processes and obstacles such as vehicle induced turbulence, street canyons, building structures, etc.

b. Sources and sinks: These can include local outdoor emissions, indoor emissions, surface deposition, etc.

c. Transformation processes: These can include local outdoor as well as indoor gas and aerosol phase chemistry, such as formation of secondary organic and inorganic aerosols.

Examples of the above are discussed next, specifically for outdoor and for indoor microenvironments.

AX3.7.4.1 Characterization of Outdoor Microenvironments

Empirical regression analyses have been used in some studies to relate specific outdoor locations - that can be interpreted as generalized types of exposure microenvironments - to spatial variability of NO_2 concentrations. For example, Gilbert et al. (2005) in May 2003 measured NO_2 for 14 consecutive days at 67 sites across Montreal, Canada. Concentrations ranged from 4.9 to 21.2 ppb (median 11.8 ppb), and they used linear regression analysis to assess the association between logarithmic values of NO_2 concentrations and land-use variables via a geographic information system. In univariate analyses, NO_2 was negatively associated with the area of open space and positively associated with traffic count on nearest highway, the length of highways within any radius from 100 to 750 m, the length of major roads within 750 m, and population density within 2000 m. Industrial land-use and the length of minor roads showed no association with NO_2. In multiple regression analyses, distance from the nearest highway, traffic count on the nearest highway, length of highways and major roads within 100 m, and population density showed significant associations with NO_2. The authors of that study point out the value of using land-use regression modeling to assign exposures in large-scale epidemological studies. Similar analyses have been performed in a predictive setting by Sahsuvaroglu et al. (2006) for Hamilton, Ontario, Canada.

The category of parameterized mass balance models for outdoor microenvironments includes various local roadway, intersection, and street canyon models. For example, Fraigneau et al. (1995) developed a simple model to account for fast nitrogen oxide – ozone reaction/dispersion in the vicinity of a motorway. Venegas and Mazzeo (2004) applied a combination of simple point and area source analytical plume models to characterize NO_2 concentration patterns in Buenos Aires, Argentina, which they used for a simplified (potential) population exposure study. ROADWAY-2 (Rao, 2002), is another near-highway pollutant dispersion model that incorporates vehicle wake parameterizations derived from canopy flow theory and wind tunnel measurements. The atmospheric velocity and turbulence fields are adjusted to account for velocity-deficit and turbulence production in vehicle wakes and a turbulent kinetic energy closure model of the atmospheric boundary layer is used to derive the mean velocity, temperature, and turbulence profiles from input meteorological data.

In parameterized street canyon models, typically, concentrations of exhaust gases are calculated using a combination of a plume model for the direct contribution and a box model for the recirculating part of the pollutants in the street. Parameterization of flow and dispersion conditions in these models is usually deduced from analysis of experimental data and model tests that considered different street configurations and various meteorological conditions.

An example of a current model that belongs in the parameterized mass balance category is the Danish Operational Street Pollution Model (OSPM) (Berkowicz, 2002), which updates earlier formulations of street canyon models such as STREET of Johnson et al. (1973) and CPBM (Canyon Plume-Box Model) of Yamartino and Weigand (1986). A variation of this simple approach is the model of Proyou et al. (1998), which uses a three-layer photochemical box model to represent a street canyon.

A variety of CFD based street canyon models have been developed in recent years (see, e.g., the series of International Conferences on Harmonization - http://www.harmo.org), employing various alternatives for closure of the turbulent transport equations. A review and intercomparison of five of these models (CHENSI, CHENSI-2, MIMO, MISKAM, TASCflow) vis-a-vis field data from a street canyon in Hannover, Germany can be found in the articles by Sahm et al. (2002) and by Ketzel et al. (2002).

These complex localized models could be useful for improving population exposure model estimates by calculating pollutant concentrations at the microenvironmental level. Lack of input parameter data and parameter variation across the modeling domain (spatial and temporal) contributes to uncertainty in microenvironmental concentrations calculated by exposre models. In such cases, parameterized mass balance models could provide outdoor concentration values for estimating exposure. If infiltration factors are known, these concentrations could also be used to estimate indoor exposures.

AX3.7.4.2 Characterization of Indoor Microenvironments

Numerous indoor air quality modeling studies have been reported in the literature; however, depending on the modeling scenario, only few of them address (and typically only a limited subset of) physical and chemical processes that affect photochemical oxidants indoors (Nazaroff and Cass, 1986; Hayes, 1989, 1991; Freijer and Bloemen, 2000).

It is beyond the scope of the present discussion to review in detail the current status of indoor air modeling. Existing indoor air concentration models indeed are available as a wide

range of (a) empirical regression relationships, (b) parameterized mass balance models (that can be either single-zone—that is, single well-mixed room—or multi-zone models), and (c) CFD formulations. Recent overviews of this area can be found in Milner et al. (2005), who focus, in particular, on the issue of entrainment from outdoor sources, and in Teshome and Haghighat, (2004), who focus on different formulations of zonal models and on how they compare with more complex CFD models.

Few indoor air models have considered detailed nonlinear chemistry, which, however, can have a significant effect on the indoor air quality, especially in the presence of strong indoor sources (e.g., gas stores and kerosene heaters, in the case of NO_2). Indeed, the need for more comprehensive models that can take into account the complex, multiphase processes that affect indoor concentrations of interacting gas phase pollutants and particulate matter has been recognized and a number of formulations have appeared in recent years. For example, the Exposure and Dose Modeling and Analysis System (EDMAS) (Georgopoulos et al., 1997c) included an indoor model with detailed gas-phase atmospheric chemistry to estimate indoor concentrations resulting from penetration and reaction of ambient pollutants. This indoor model was dynamically coupled with (a) the outdoor photochemical air quality models UAM-IV and UAM-V, which provided the gas-phase composition of influent air; and (b) with a physiologically based uptake and dosimetry model. Subsequent work (Isukapalli et al., 1999) expanded the approach of the EDMAS model to incorporate alternative representations of gas-phase chemistry as well as multiphase photochemistry and gas/aerosol interactions. The microenvironmental model corresponding to this more general formulation is mathematically represented by the following equation, when an assumption of uniform mixing is used for each component (e.g., individual room) of the indoor environment. Sarwar et al. (2001) presented a more comprehensive modeling study of the gas phase aspects of ozone indoor chemistry focusing on the impact of different factors (such as outdoor ozone, indoor emissions, ventilation rates, etc.) on the levels of indoor hydroxyl radicals (OH), which in turn are expected to control the rate of formation of secondary toxicants indoors.

$$V_i \frac{dC_i^{(m)}}{dt} = \sum_{i-1}^{N} Q_{ji} C_j^{(m)} - \sum_{j=1}^{N} Q_{ij} C_i^{(m)} + S_i^{(m)} + \sum_{j=1}^{N} K_{ij}^{(m)} a_{ij} (C_{ij}^{*(m)} - C_i^{(m)}) + R_i^{(m)} \quad \text{(AX3-19)}$$

where,

V_i = volume of compartment (m^3)

C_i = concentration of species in compartment (mol/m^3)

K_{ij} = mass transfer coefficient from compartment (m/h)

a_{ij} = interfacial air exchange area between compartments (m^2)

C_{ij} = concentration in compartment i in equilibrium with concentration in j (mol/m^3)

Q_{ij} = volumetric flow rate from compartment i to j (m^3/h)

R_i = rate of formation of species in compartment i (gmol/h)

and,

$$S_i \begin{cases} S_{i,emis} - S_{i,depos} - S_{i,condens} & ;for\ gases \\ S_{i,emis} - S_{i,depos} + S_{i,resusp} + S_{i,condens} + S_{i,nucl} + S_{i,coag;for\ PM} \end{cases}$$

More recent work (Sørensen and Weschler, 2002) has coupled CFD calculations with gas-phase atmospheric chemistry mechanisms to account for the impact of nonideal flow mixing (and associated concentration gradients) within a room on the indoor spatial distribution of ozone and other secondary pollutants. This work has identified potential limitations associated with the assumption of uniform mixing in indoor microenvironments when calculating personal exposures.

A recent indoor air model that specifically focuses on NO_2 (along with CO, PM_{10}, and $PM_{2.5}$ is INDAIR (Dimitroulopoulou et al., 2006). The INDAIR model considers three interconnected residential microenvironments: kitchen, lounge, and bedroom. Removal processes are lumped together and quantified via an apparent deposition velocity. Specifically, a loss rate of $0.99 \pm 0.19\ h^{-1}$ (Yamanaka, 1984), is used in this model corresponding to a mean deposition velocity of 1.2×10^{-4} m s^{-1}. The sources of NO_2 considered in INDAIR are from gas

stove cooking and from cigarette smoking, but only the former contributes significantly to indoor NO_2 levels, based on available model parameterizations.

Estimation of NO_2 emission rates from gas cooking utilized the following empirical information: (a) NO_x emission rate equal to 0.125 g kWh^{-1} (Wooders, 1994); (b) an assumption that NO_2 represents 25% of the total NO_x emissions and (c) gas consumption per household in cooking equal to 5–7 kWh day^{-1}, assuming 1 h cooking per day. By multiplying the estimates in (a), (b), and (c) together, NO_2 gas cooking emission rates were calculated to be in the range 0.16 to 0.22 g h^{-1}, with a uniform distribution.

In a range of simulations performed with INDAIR for houses in the UK, it was found that the predicted maximum 1-h mean concentrations in the kitchen were increased, compared to no-source simulations, by a factor of 10 for NO_2 (30 for PM_{10} and 15 for $PM_{2.5}$) and were higher in winter than in summer. Cooking activity in the kitchen resulted in significantly elevated 24 h mean concentrations of NO_2, PM_{10}, and $PM_{2.5}$ in the lounge, as well as the kitchen, while there was a relatively small effect in the bedroom, which was not connected directly to the kitchen in the model structure (i.e., the direct internal air exchange rate was zero).

A very wide range of predictions was derived from the INDAIR simulations. The 95th percentile concentrations were typically 50% higher than mean concentrations during periods of average concentration, and up to 100% higher than mean concentrations during concentration peaks, which were associated with cooking emissions. There was approximately a factor of 2 variation in concentrations, and all modeled concentrations were below those outdoors. The effect of cooking was to shift the distribution to the right, but the degree of variation was not greatly increased. This may reflect the fact that for the fixed emission scenarios that were used, the additional variation in emission rates was small compared to that of other factors such as deposition rate and air exchange rate. In this scenario, modeled concentrations in the lounge all remained below those outdoors, but a proportion of kitchens (16%) had modeled values above the outdoor concentration. For the gas-cooking scenario, indoor/outdoor ratios for NO_2 ranged from 0.5 to 0.8 for the bedroom, 0.7 to 1.6 for the lounge and 0.9 to 3.6 for the kitchen. According to Dimitrolopoulou et al. (2006), these results were broadly consistent with indoor/outdoor ratios reported for the UK. Modeled peak concentrations associated with gas cooking, of about 300 ppb in the kitchen and 100 ppb in the lounge, were also consistent with results from UK studies.

AX3.7.4.3 Characterization of Activity Events

An important development in inhalation exposure modeling has been the consolidation of existing information on activity event sequences in the Consolidated Human Activity Database (CHAD) (McCurdy, 2000; McCurdy et al., 2000). Indeed, most recent exposure models are designed (or have been re-designed) to obtain such information from CHAD which incorporates 24-h time/activity data developed from numerous surveys. The surveys include probability-based recall studies conducted by Environmental Protection Agency and the California Air Resources Board, as well as real-time diary studies conducted in individual U.S. metropolitan areas using both probability-based and volunteer subject panels. All ages of both genders are represented in CHAD. The data for each subject consist of one or more days of sequential activities, in which each activity is defined by start time, duration, activity type (140 categories), and microenvironment classification (110 categories). Activities vary from one min to one h in duration, with longer activities being subdivided into clock-hour durations to facilitate exposure modeling. A distribution of values for the ratio of oxygen uptake rate to body mass (referred to as metabolic equivalents or METs) is provided for each activity type listed in CHAD. The forms and parameters of these distributions were determined through an extensive review of the exercise and nutrition literature. The primary source of distributional data was Ainsworth et al. (1993), a compendium developed specifically to facilitate the coding of physical activities and to promote comparability across studies.

AX3.7.4.4 Characterization of Inhalation Intake and Uptake

Use of the information in CHAD provides a rational way for incorporating realistic intakes into exposure models by linking inhalation rates to activity information. As mentioned earlier, each cohort of the pNEM-type models, or each (virtual or actual) individual of the SHEDS, MENTOR, APEX, and HAPEM4 models, is assigned an exposure event sequence derived from activity diary data. Each exposure event is typically defined by a start time, a duration, assignments to a geographic location and microenvironment, and an indication of activity level. The most recent versions of the above models have defined activity levels using the activity classification coding scheme incorporated into CHAD. A probabilistic module within these models converts the activity classification code of each exposure event to an energy expenditure rate, which in turn is converted into an estimate of oxygen uptake rate. The oxygen uptake rate is then converted into an estimate of total ventilation rate (V_E), expressed in liters

min^{-1}. Johnson (2001) reviewed briefly the physiological principles incorporated into the algorithms used in pNEM to convert each activity classification code to an oxygen uptake rate and describes the additional steps required to convert oxygen uptake to V_E.

McCurdy (1997a,b, 2000) has recommended that the ventilation rate should be estimated as a function of energy expenditure rate. The energy expended by an individual during a particular activity can be expressed as EE = (MET)(RMR) in which EE is the average energy expenditure rate (kcal min^{-1}) during the activity and RMR is the resting metabolic rate of the individual expressed in terms of number of energy units expended per unit of time (kcal min^{-1}). MET (the metabolic equivalent of tasks) is a ratio specific to the activity and is dimensionless. If RMR is specified for an individual, then the above equation requires only an activity-specific estimate of MET to produce an estimate of the energy expenditure rate for a given activity. McCurdy et al. (2000) developed distributions of MET for the activity classifications appearing in the CHAD database.

Finally, in order to relate intake to dose delivered to the lungs, it is important to take into account the processes affecting uptake following inhalation intake of NO_2, in a biologically based dosimetry modeling framework. As a reactive gas, NO_2 participates in transformation reactions in the lung epithelial lining fluid, and products of these reactions are thought to be responsible for toxic effects (Postlethwait et., 1991), although kinetic modeling of these reactions has not been performed. Dosimetry models indicate that deposition varies spatially within the lung and that this spatial variation is dependent on ventilation rate (Miller et al., 1982; Overton and Graham, 1995). Controlled exposure studies found that fractional uptake of NO_2 increases with exercises and ventilation rate (e.g., Bauer et al., 1986), making activities with high MET values important for quantifying total NO_2 exposure. Further discussion of NO_2 dosimetry modeling is provided in Section 4.2.

AX3.7.5 Concluding Comments

An issue that should be mentioned in closing is that of evaluating comprehensive prognostic exposure modeling studies, for either individuals or populations, with field data. Although databases that would be adequate for performing a comprehensive evaluation are not expected to be available any time soon, there have been a number of studies, reviewed in earlier sections of this Chapter, that can be used to start building the necessary information base. Some

of these studies report field observations of personal, indoor, and outdoor ozone levels and have also developed simple semi-empirical personal exposure models that were parameterized using the observational data and regression techniques.

In conclusion, though existing inhalation exposure modeling systems have evolved considerably in recent years, limitations of available modeling methods and data, in relation to potential NO_2 studies that include the following, should be taken into account and be addressed by future research efforts:

- Ambient photochemical modeling systems are not optimized for estimating NO_2 at a local scale.
- Subgrid scale modeling (LES, RANS, DNS) is needed to properly characterize effects of nonhomogeneous mixing (i.e., of spatial subgrid variability) on fast nonlinear chemical transformations; the outcomes of this characterization then should be incorporated in simpler models, appropriate for use in conjunction with exposure modeling systems.
- Microenvironmental modeling efforts need to balance mechanistic detail and usability by developing:
  - — A simplified but adequate indoor chemistry mechanism for NO_2 and related oxidants,
  - — Databases of realistic distributions of indoor NO_2 source magnitudes and activities,
  - — Flexible, multi-zonal models of indoor residential and occupational microenvironments.

Existing prognostic modeling systems for inhalation exposure can in principle be directly applied to, or adapted for, NO_2 studies; APEX, SHEDS, and MENTOR-1A are candidates. However, such applications would be constrained by data limitations such as ambient characterization at the local scale and by lack of quantitative information for indoor sources and sinks.

AX3.8 EXPOSURE ERROR

Discussions in this section focus on the errors associated with exposure assessments and, in particular, with those that may be associated with using ambient NO_2 as a surrogate of personal NO_2 exposure in epidemological time series studies. As shown in Figure AX3.24, exposure error is one of the errors associated with epidemological studies linking pollutant

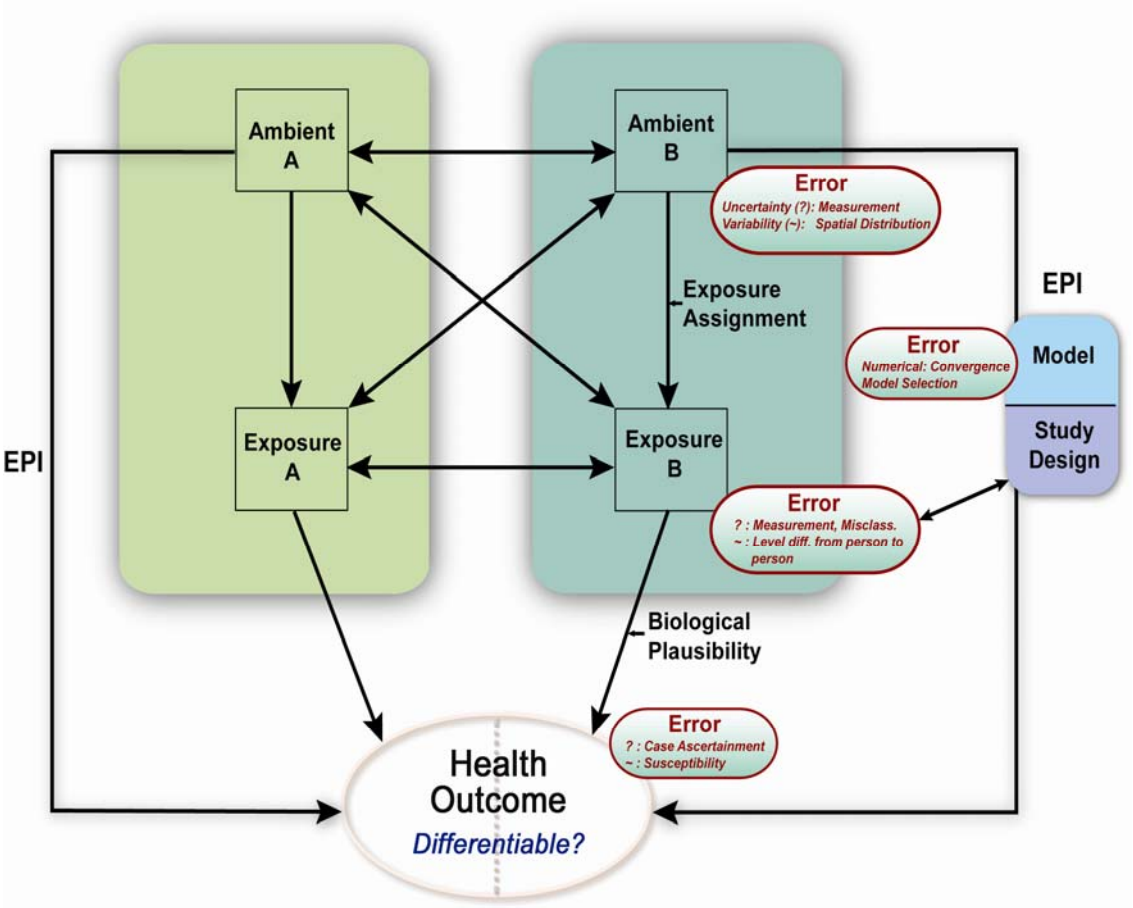

Figure AX3.24. Errors associated with components of the continuum from ambient air pollution to adverse health outcome.

concentrations in ambient air and human health responses. How exposure errors influence the epidemological findings depend upon the design of the epidemological study. In this section, the exposure errors will be discussed in the context of two common environmental epidemological study designs, time-series studies and chronic studies, in which central site NO_2 concentrations are used as surrogates of personal exposure.

In a broader sense, NO_2 is an indicator of a chemical mixture, which might be the real agent(s) leading to epidemological findings. Ambient, indoor or personal NO_2 might indicate different chemical mixtures because of differences in the infiltration efficiency or chemical reactivity of other NO_y species or in the composition of nearby sources. When using ambient NO_2 as a surrogate of personal exposure, issues of confounding and surrogate are raised.

Confounding issues have been discussed in Section AX3.6. A brief summary of the confounding issues and a brief discussion of the surrogate issues will be provided in this section.

Usually when discussing errors in the context of exposure assessments, errors resulting from limitations of analytical capabilities of monitoring instruments are lumped together with those caused by environmental factors such as spatial heterogeneity in ambient concentrations, the lack of identification of indoor and neighborhood sources etc. In certain instances these different errors may be linked.

Measurements of NO_2 are subject to artifacts both at the ambient level and at the personal level. A discussion of the errors associated with ambient monitors is given in Section 2.8, and one for errors associated with personal monitors is given in Section AX3.4. As noted earlier, measurements of ambient NO_2 are subject to variable interference caused by other NO_y compounds, in particular PANs, organic nitrates, particulate nitrate and HNO_2 and HNO_3. The latter is taken up on inlet walls to varying degrees and likely causes variable (positive) artifacts in NO_2 measurements.

Personal monitors are subject to interference by SO_2 and HONO and it is not clear to what extent they are affected by interference by the NO_y species interfering with the ambient monitors. In addition, personal monitors generally require longer sampling times (typically from about a day to two weeks) and so will not be able to identify peak exposures occurring on time scales of a few hours or less. As noted by Pilotto et al. (1997) these exposures would have been averaged out and associated health outcomes would not be properly attributed by monitors requiring longer sampling times. Often personal concentrations may either be below or not very much above detection limits for the most commonly used personal samplers (see Table AX3.6). Thus, associations between ambient and personal concentrations could be weakened between ambient and personal concentrations of a given pollutant. In studies of multiple pollutants, personal concentrations of one pollutant may be more strongly associated with ambient concentrations of another pollutant if the measurements of the latter at the personal level are subject to larger analytical errors than are measurements of the former at the personal level.

Spatial heterogeneity in ambient concentrations helps determine how well concentrations measured at ambient monitoring sites reflect exposures at the community and personal levels. Correlations between different pairs of monitoring sites are not sufficient for characterizing spatial variability, as there may be significant differences in concentrations among monitoring

sites. This point has been demonstrated in Chapter 3 the latest AQCD for PM (U.S. Environmental Protection Agency, 2004) and Chapter 3 the latest AQCD for ozone and other photochemical oxidants (U.S. Environmental Protection Agency, 2006a). As described earlier in Section AX3.2, concentrations of NO_2 are highly variable across the urban areas examined and will result in exposure characterization errors at least as significant as, if not larger, than those for O_3 and $PM_{2.5}$. The problem is exacerbated for NO_2 because of the sparseness of NO_x monitors, compared to monitors for PM and O_3. Thus, the use of central site monitors may be more problematic for NO_2 than for $PM_{2.5}$ (e.g.). As a result, little relation might be found between ambient central site monitors and personal exposures and/or indoor concentrations and stronger associations might be found between cross pollutants at the ambient and personal levels. In this case, it may be necessary to supplement existing ambient measurements to derive ambient concentrations that are consistent with those of other pollutants, e.g., by the use of supplemental 'outdoor' monitors. Additional complexity arises if horizontal spatial gradients are large enough, as might happen in going from urban to rural environments, as the lowest values measured might be beneath quantification limits or even beneath detection. Small scale horizontal variability especially as found near roads could be large.

As noted earlier in Section AX3.2, variability in the vertical must be considered in addition to horizontal variability. NO_2 emitted at or near ground level exhibits strong vertical gradients. Restrepo et al. (2004) found that NO_2 measured at 15 m above the surface was a factor of higher than measurements of NO_2 at 4 m. Monitors placed at heights such as these will be found in many inner urban areas.

In the framework developed by Zeger (2000) for analyzing errors in time-series epidemological studies associated with exposure measurement errors, exposure errors could be classified into three components: (1) the difference between true ambient concentration and the measured ambient concentration, (2) the difference between the measured ambient concentration and the community ambient exposure, and (3) the difference between the community ambient exposure and the personal ambient exposure. These differences mentioned above are determined by (1) the reliability of measurement techniques, (2) the spatial and temporal variation of ambient NO_2 concentrations, and (3) personal activity and microenvironment characteristics.

In the context of chronic epidemological studies, the issue of misclassification also arises. Personal exposure is composed of exposures to both ambient sources and nonambient sources. If

total personal NO_2 exposure is assumed to be responsible for the observed health outcomes, the use of ambient concentration as a surrogate for personal exposure could lead to misclassification and bias the epidemological findings. The degree of the misclassification also depends on the spatial and temporal variation of ambient NO_2, personal activities and microenvironment characteristics.

In the Danish children exposure study, front door NO_2 as well as personal NO_2 concentrations were measured (Raaschou-Nielsen et al., 1997). To evaluate the extent of misclassification using outdoor NO_2 as an indicator of personal exposure, Raaschou-Nielsen et al. (1997) reported that both the sensitivity (the proportion of correctly classified highly exposure) and the specificity (the proportion of correctly classified low exposure) were 81% in Copenhagen and 74% in the rural areas. Similar results were reported by Lee, et al., (2004).

Exposure measurement errors could also be evaluated by comparing the within subject and between subject variations of individual exposures. The higher the ratio of within variance and between variance, the more the true exposure-effect relationship is biased (Armstrong et al., 1992). During the Los Angeles NO_2 exposure study, Spengler et al. (1994) reported that the within personal variation was 61.2 $\mu g/m^3$ and the variation between personal exposure was 608.2 $\mu g/m^3$. Alm et al. (1998) reported that within personal variation explained 59% of the total personal exposure variation and 41% of the total variation was accounted by between-subject variation.

Simply speaking, two parameters could be used to evaluate the feasibility of using ambient NO_2 concentrations as a surrogate for personal exposure: the correlation coefficient between personal exposure and ambient concentrations (especially in the context of longitudinal design and daily-averaged design), and the difference between personal exposure and ambient concentration. Extensive discussions of this issue have been provided in Section AX3.5, such discussions are not repeated here and only general conclusions will be provided. The correlation between personal exposure and ambient concentrations range from moderate to good. Personal exposure concentrations are generally lower than ambient concentrations for homes with no indoor or local sources but higher than ambient concentration for homes with indoor or local sources.

In a broader context, NO_2 serves as an indicator of a pollutant mixture whose components have different physical and chemical properties that may be the real agent(s) causing the adverse

health effects. The components of the mixture are either primary or secondary, i.e., they either come from direct emissions or form through atmospheric chemical reactions. When the ambient mixture infiltrates into microenvironments, some components are lost due to absorption and chemical reaction, while some new components are formed through chemical reactions in indoor air. At the same time, indoor primary sources could add more NO_2 along with other pollutants in the indoor environments. When evaluating the question of whether ambient NO_2 is the agent causing the observed adverse health effects, the two issues of confounding and surrogacy are raised.

The definition and discussion of the confounding issue from the perspective of exposure analysis could be found in Section AX3.6. In Section AX3.6, the following five questions were evaluated (the five arrows in Figure AX3.24): (1) Are ambient copollutant concentrations significantly associated with ambient NO_2? (2) Are personal exposures to copollutants significantly associated with personal exposures to NO_2? (3) Are ambient pollutant concentrations associated with their respective personal exposures? (4) Are ambient copollutants surrogates for personal exposure to NO_2? (5) Is ambient NO_2 a surrogate for personal exposure to copollutants? Based on the fact that NO_2 is correlated with other copollutants at both ambient level and personal exposure levels and that cross-level correlations were also observed, we concluded that caution should be exercised when dealing with the observed NO_2 health effect and a more comprehensive analysis should be performed in conjunction with other components of the risk assessment.

Another issue raised is the surrogate issue. There are different meanings associated with, to use the word "surrogate". In summary, there are three scenarios involving the concept of a surrogate and each one is associated with a question: (1) At ambient level, is ambient NO_2 a good surrogate (tracer) for some ambient chemical or chemical mixture? (2) At personal exposure levels, is personal NO_2 exposure a good surrogate (tracer) for some chemical or chemical mixture of personal exposure? and (3) At health effect levels, is NO_2 a good surrogate for some chemical or chemical mixture causing an adverse health outcome? The first two questions could be sufficiently answered by various source apportionment approaches to evaluate the co-variation of NO_2 with other pollutants. The third question is evaluated in Figure AX3.25 with a systematic approach considering biological plausibility and exposure assessment.

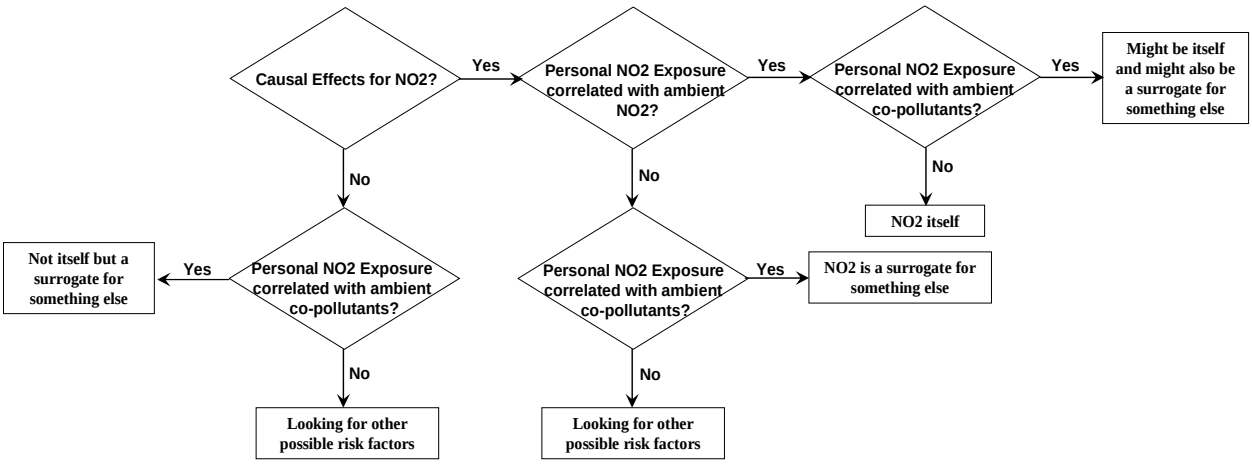

Figure AX3.25. A systematic approach to evaluate whether NO_2 itself is causing the observed adverse health outcome or NO_2 is acting as a surrogate for other pollutants.

TABLE AX3.1. SUMMARY OF PERCENTILES OF NO_2 DATA POOLED ACROSS MONITORING SITES (2003-2005) CONCENTRATIONS ARE IN PPM

Pooled Group/ Avg Time	Number of Values	Mean	Percentiles											
			1	5	10	25	30	50	70	75	90	95	99	Max
1-h Max Concentrations														
Monitors in CMSAs	288008	0.029	0.003	0.007	0.010	0.017	0.019	0.027	0.036	0.038	0.048	0.055	0.072	0.201
Monitors not in CMSAs	460913	0.008	0.001	0.001	0.001	0.002	0.003	0.005	0.009	0.010	0.019	0.026	0.040	0.189
1-h Avg. Concentrations														
Monitors in CMSAs	6163408	0.015	0.001	0.003	0.003	0.006	0.007	0.012	0.019	0.022	0.033	0.040	0.053	0.201
Monitors not in CMSAs	460913	0.008	0.001	0.001	0.001	0.002	0.003	0.005	0.009	0.010	0.019	0.026	0.040	0.189
Daily 24-h Avg. Concentrations														
Monitors in CMSAs	282810	0.015	0.002	0.003	0.005	0.008	0.009	0.012	0.019	0.021	0.028	0.034	0.045	0.129
Monitors not in CMSAs	20635	0.008	0.001	0.001	0.001	0.003	0.003	0.006	0.010	0.011	0.017	0.021	0.030	0.081
2-week Avg. Concentrations														
Monitors in CMSAs	21779	0.015	0.003	0.005	0.006	0.009	0.010	0.014	0.019	0.020	0.026	0.031	0.038	0.076
Monitors not in CMSAs	1588	0.008	0.001	0.001	0.001	0.003	0.003	0.007	0.009	0.012	0.016	0.020	0.030	0.039
Yearly Avg. Concentrations														
Monitors in CMSAs	758	0.015	0.004	0.006	0.007	0.011	0.012	0.015	0.018	0.019	0.025	0.028	0.033	0.037
Monitors not in CMSAs	51	0.008	0.001	0.001	0.002	0.003	0.005	0.009	0.012	0.012	0.015	0.016	0.017	0.017
3-yr Avg. Concentrations														
Monitors in CMSAs	247	0.015	0.004	0.006	0.007	0.011	0.012	0.015	0.018	0.019	0.025	0.028	0.032	0.033
Monitors not in CMSAs	15	0.008	0.001	0.001	0.002	0.003	0.006	0.008	0.012	0.012	0.014	0.016	0.016	0.016

TABLE AX3.2. SPATIAL VARIABILITY OF NO_2 IN SELECTED UNITED STATES URBAN AREAS

	Mean 1-h Concentration(ppb)	r	P90 (ppb)	COD
New York, NY (5)	29 (25 – 37)	0.77-0.90	7 – 19	0. 08 – 0.23
Atlanta, GA (5)	11 (5 – 16)	0.22-0.89	7 – 24	0.15 – 0.59
Chicago, IL (7)	22 (6 – 30)	-0.05 – 0.83	10 -39	0.13 – 0.66
Houston, TX (7)	13 (7 – 18)	0.31 – 0.80	6 – 20	0.13 – 0.47
Los Angeles, CA (14)	25 (14- 33)	0.01 – 0.90	8 – 32	0.08 – 0.51
Riverside, CA (9)	21 (5 – 32)	0.03 – 0.84	10 – 40	0.14 – 0.70

TABLE AX3.3. NO_X AND NO_Y CONCENTRATIONS AT REGIONAL BACKGROUND SITES IN THE EASTERN UNITED STATES. CONCENTRATIONS ARE GIVEN IN PPB

	Shenandoah NP, VA	Harvard Forest, MA
NO		
Winter	0.39-2.2 [1]	—
Summer	0.12-0.28	—
NO_x	—	—
Winter	—	1-15
Summer	—	0.4-1.2
NO_y	—	—
Winter	2.7-8.6	4.4 [2]
Summer	2.3-5.7	2.7 [2]

[1] Ranges represent 1σ limits.

[2] Values represent medians.

TABLE AX3.4. RANGE OF PEARSON CORRELATION COEFFICIENTS BETWEEN NO_2 AND O_3, CO AND $PM_{2.5}$

Monitoring Sites in Selected Areas	Copollutant		
	O_3	CO	$PM_{2.5}$
Los Angeles, CA	-0.59 to 0.19	0.11 to 0.83	0.45 to 0.56
Riverside, CA	-0.26 to 0.28	0.15 to 0.65	—
Chicago, IL	-0.20 to -0.13	-0.10 to 0.53	0.21 to 0.49
Washington, DC	—	—	—
New York City, NY	—	—	—

TABLE AX3.5. PASSIVE SAMPLERS USED IN NO_2 MEASUREMENTS

Passive Sampler	Dimension (diffusion length × cross-sectional area)	Absorbent	Analytical Method	Sampling Rate		Reference
				Manufacturer	Experiment	
Palmes tube	7.1cm × 0.71cm^2	Triethanolamine	Spectrophotometry	N.A.	0.92 cm^3/min	Palmes et al. (1976)
Gradko sampler	7.1cm × 0.93cm^2	Triethanolamine	Spectrophotometry	1.2 cm^3/min	1.212 cm^3/min	Plaisance et al. (2004) Gradko (2007)
Passam sampler Short	0.74cm × 0.75cm^2	Triethanolamine	Spectrophotometry	15.5 cm^3/min	N.A.	Passam (2007)
Passam sampler Long				0.854 cm^3/min	0.833 cm^3/min	
Analyst™	2.54cm × 3.27cm^2	Active charcoal	Gas chromatography	N.A.	12.3 cm^3/min	De Santis et al. (2002)
Yanagisawa badge	1.0cm × 20cm^2	Triethanolamine	Spectrophotometry	N.A.	N.R.	Yanagisawa et al. (1982
Ogawa sampler	0.6cm × 0.79cm^2	Triethanolamine	Spectrophotometry	N.A.	16.2 cm^3/min	Ogawa & Company (1998[a]) Gerboles et al. (2006[a])
IVL sampler	1.0cm × 3.14cm^2	Potassium iodide & sodium arsenite	Spectrophotometry	N.A.	29 cm^3/min	Ferm et al. (1998)
Willems badge	0.6cm × 5.31cm^2	Triethanolamine-acetone	Spectrophotometry	N.A.	46 cm^3/min	Hagenbjörk-Gustafsson et al. (2002)
Radiello®	1.8cm × 2.0cm^2	Triethanolamine	Spectrophotometry	75 cm^3/min	N.R.	Radiello® (2006)
EMD sampler	N.A.	Triethanolamine	Ion chromatography	N.A.	53.4 cm^3/min	Piechocki-Minguy et al. (2006)

*N.A.: not available; N.R.: not reported.

TABLE AX3.6. THE PERFORMANCE OF SAMPLER/SAMPLING METHOD FOR NO_2 MEASUREMENTS IN THE AIR

Type	Sampler		Optimal Duration of Sampling	Concentration Range	Detection Limit	Comment
Active sampling	Impinger method		2-24 h	10 – 400 ppb	0.2 ppb	
	Chemiluminescence		Continuous	0.5 – 1000 ppb	0.05 ppb	RSD < 5%
	Personal monitor		Real-time	0.1 – 50 ppm	0.1 ppm	Accuracy ± 5%
Passive sampling	Palmes tube		1-4 wks	10 - 100 ppb	10 ppb	
	Gradko sampler		2-4 wks	1.0 – 10,000 ppb	0.5 ppb	Precision ± 5% above 5 ppb
	Passam sampler	Short	8-48 h	5 – 240 µg/m^3	2-5 µg/m^3	Uncertainty ~ 27% at 80 µg/m^3
		Long	1-4 wks	1 – 200 µg/m^3	0.64 µg/m^3	Uncertainty ~ 25% at 20-40 µg/m^3
	Analyst™		1-3 mos	24 – 1,237 µg/m^3	100 µg/m^3	Accuracy ± 5%; Precision within 3%
	Yanagisawa badge		1-14 days	N.R.	3.0 ppb	
	Ogawa sampler		24-168 h	0 – 3,600 ppb	2.3 ppb	
	IVL sampler		1 mo +	0.1 – 400 µg/m^3	0.1 µg/m^3	RSD ~ 4%
	Willems badge		2-8 h & 1-7 days	2.0 – 150 µg/m^3	2 µg/m^3	Uncertainty ~ 24%; RSD 22%
	Radiello®		1-24 h & 1-7 days	1.0 – 496 ppb	1.0 ppb	Uncertainty ~ 12%
	EMD sampler		1-24 h	N.R.	11 µg/m^3	Uncertainty ~ 28%

N.R.: not reported.

TABLE AX3.7. NO_2 CONCENTRATIONS (PPB) IN HOMES IN LATROBE VALLEY, VICTORIA, AUSTRALIA

	Living Room			Kitchen		
	Mean ppb	**Min ppb**	**Max ppb**	**Mean ppb**	**Min ppb**	**Max ppb**
No source	3.77	< 0.37	9.27	3.82	< 0.37	8.17
Gas stove only	6.70	1.57	18.32	8.01	2.62	24.14
Gas heater only	6.86	2.20	18.06	7.33	2.88	26.23
Smoking only	6.02	0.94	14.61	6.60	1.83	16.44
Multiple sources	14.50	2.25	114.66	10.73	2.62	128.80

Source: Garrett et al. (1999).

TABLE AX3.8. NO_2 CONCENTRATIONS (PPB) IN HOMES IN CONNECTICUT

	No Gas Stove Used in Monitoring Period						Yes Gas Stove Used in Monitoring Period					
Secondary Heating Source	**N**	**10th**	**25th**	**Median**	**75th**	**90th**	**N**	**10th**	**25th**	**Median**	**75th**	**90th**
None	1018	1.7	3.5	**6.3**	12.3	28.2	564	8.4	14.5	**22.7**	33.8	48.1
Gas space heater	6	0.1	9.2	**15.3**	68	69.6	6	19.5	34.6	**36.6**	54.8	147.2
Wood burning source	200	1.8	3.6	**5.9**	12.2	28.2	78	6	9.5	**16.7**	31.4	58.6
Kerosene heater	159	3.3	7.1	**18.9**	42.7	88.3	14	0	9.6	**17.2**	33.6	46.1
GSH + Wood	3	12.6	12.6	**80.6**	81.9	81.9	5	36.2	44.8	**57.1**	114.2	156.6
GSH + KH	0	--	--	--	--	--	1	n/a	n/a	**147.7**	n/a	n/a
Wood + KH	73	1.9	8.2	**16.4**	35.2	66.8	5	8.9	12.7	**17.3**	23.5	72.9
GSH + Wood + KH	0	--	--	--	--	--	1	n/a	n/a	**107.8**	n/a	n/a

Source: Triche et al. (2005).

TABLE AX3.9. NO_2 CONCENTRATIONS NEAR INDOOR SOURCES – SHORT-TERM AVERAGES

Average Concentration (ppb)	Peak Concentration (ppb)	Comment	Reference
191 kitchen 195 living room 184 bedroom	375 kitchen 401 living room 421 bedroom	Cooked full meal with use of gas stove and range for 2 h, 20 min; avg conc. is time-weighted over 7 h.	Fortmann et al. (2001)
400 kitchen, living room, bedroom	673 bedroom	Automatic oven cleaning of gas stove. Avgs are over the entire cycle.	Fortmann et al. (2001)
90 (low setting) 350 (med setting) 360 (high setting)	N/R[1]	Natural gas unvented fireplace,[2] 2-h-time-weighted avg in main living area of house (177 m^3).	Dutton et al. (2001)
N/R	1000	Room concentration with kerosene heater operating for 46 min.	Girman et al. (1982)
N/R	1500	Room concentration with gas heater operating for 10 min.	Girman et al. (1982)
180 to 650	N/R	Calculated steady-state concentration from specific unvented gas space heaters operating in a 1400 ft^2 house, 1.0 ach.	Girman et al. (1982)

[1] N/R = Not Reported.

[2] Unvented fireplaces are not permitted in many areas such as California.

Source: Adapted from CARB (2007).

TABLE AX3.10. NO_2 CONCENTRATIONS NEAR INDOOR SOURCES – LONG-TERM AVERAGES

Average Concentration (ppb)	Comment	Reference
30 to 33	Gas stoves with pilot lights.	Lee et al. (1998)
22	Gas stoves without pilot lights.	
6 to 11	Electric ranges. Study conducted in 517 homes in Boston, values represent 2-wk avgs.	
55 (Median)	Gas space heaters.	Triche et al. (2005)
41 (90th %-ile)	No indoor combustion source.	
80 (90th %-ile)	Fireplaces.	
84 (90th %-ile)	Kerosene heater.	
147 (90th %-ile)	Gas space heaters.	
52 (90th %-ile)	Wood stove.	
	All values represent 2-wk avgs in living rooms.	
18 bedrooms 19 living rooms 15 outdoors	Almost all homes had gas stoves. Values represent 2-wk avgs.	Zipprich et al. (2002)

TABLE AX3.11. SUMMARY OF REGRESSION MODELS OF PERSONAL EXPOSURE TO AMBIENT/OUTDOOR NO_2

Study	Location	Season	Model type	Slope (SE)	Intercept / ppb	R^2
Rojas-Bracho et al. (2002)	Santiago, urban	Winter	Personal vs. outdoor	0.33 (0.05)	7.2	0.27
Alm et al. (1998)	Helsinki, downtown + suburban	Winter + Spring	Personal vs. central	0.3	5.0	0.37
			Personal vs. outdoor	0.4	4.7	0.86
Monn et al. (1998)	Four urban + two rural + two alpine	All	Personal (all subjects) vs. outdoor	0.45	7.2	0.33
			Personal (no smokers and gas cooking) vs. outdoor	0.38	7.2	0.27
Levy et al. (1998a)	15 cities in 18 countries	Winter	Personal vs. outdoor	0.49	14.5	—
Spengler et al. (1994)	Los Angeles Basin	All	Personal vs. outdoor	0.56	15.8	0.51
Sørensen et al. (2005)	Copenhagen, urban	All	Personal vs. outdoor	0.60 (0.07)	—	—
		(>8 ºC)	Personal vs. outdoor	0.68 (0.09)	—	—
		(<8 ºC)	Personal vs. outdoor	0.32 (0.13)	—	—
		All	Personal vs. central	0.56 (0.09)	—	—
Sarnat et al. (2001)	Baltimore	Summer	Personal vs. central	0.04*	9.5	—
		Winter	Personal vs. central	−0.05*	18.2	—
Sarnat et al. (2005)	Boston	Summer	Personal vs. central	0.19	—	—
		Winter	Personal vs. central	−0.03*	—	—
Sarnat et al. (2006)	Steubenville	Summer	Personal vs. central	0.25 (0.06)	—	0.14
		Fall	Personal vs. central	0.49 (0.05)	—	0.43

*Not significant at the 5% level.

TABLE AX3.12. AVERAGE AMBIENT AND NONAMBIENT CONTRIBUTIONS TO POPULATION EXPOSURE

Study	Model Type	Slope (SE)	Intercept / ppb	Mean of Personal Total Exposure / ppb	Mean Ambient Contribution / ppb	Percent Ambient Contribution %	Percent Nonambient Contribution %
Rojas-Bracho et al. (2002)	Personal vs. outdoor	0.33 (0.05)	7.2	36.4	7.2	19.8	80.2
Alm et al. (1998)	Personal vs. central	0.3	5.0	—	5.0	—	—
	Personal vs. outdoor	0.4	4.7	—	4.7	—	—
Monn et al. (1998)	Personal (all subjects) vs. outdoor	0.45	7.2	14.1	7.2	51.1	48.9
	Personal (no smokers and gas cooking) vs. outdoor	0.38	7.2	—	7.2	—	—
Levy et al. (1998a)	Personal vs. outdoor	0.49	14.5	28.8	14.5	50.3	49.7
Spengler et al. (1994)	Personal vs. outdoor	0.56	15.8	37.6	15.8	42.0	58.0

** Not reported.

TABLE AX3.13. THE ASSOCIATION BETWEEN PERSONAL EXPOSURES AND AMBIENT CONCENTRATIONS

Study	Study Design	Association Variable	Location	Season	r_p, r_s, or R^2
Linn et al. (1996)	Children, Southern California, 24 h averaged, one wk consecutive measurement for each season (fall, winter, and spring 1992-1994) for each child.	Personal vs. central	pooled	pooled	0.63 (r_p)
Krämer et al. (2000)	Children, West Germany, two one-wk averaged measurements for each child each in March and Sept 1996	Personal vs. outdoor	pooled	pooled	0.37 (r_p)
		Personal vs. outdoor	urban	pooled	0.06 (r_p)
Rojas-Bracho et al. (2002)	Children, Santiago, 24 h averaged sample for five consecutive days for each child, winters of 1998 and 1999	Personal vs. outdoor	urban	winter	0.27 (R^2)
Raaschou-Nielsen et al. (1997)	Children, Copenhagen and rural areas, one-wk averaged, 2 measurements for each child in each month (Oct 1994, April, May, and June 1995)	Personal vs. outdoor	urban	pooled	0.15 (R^2)
		Personal vs. outdoor	rural	pooled	0.35 (R^2)
Alm et al. (1998)	Children, Helsinki, one-week averaged, 13 wks for each child in each season (winter and spring 1991)	Personal vs. outdoor	downtown	winter	0.46 (r_p)
		Personal vs. outdoor	suburban	winter	0.49 (r_p)
		Personal vs. outdoor	downtown	spring	0.80 (r_p)
		Personal vs. outdoor	suburban	spring	0.82 (r_p)
		Personal vs. central	downtown	spring	0.64 (r_p)
		Personal vs. central	suburban	spring	0.78 (r_p)
		Personal vs. outdoor	pooled	pooled	0.86 (R^2)
		Personal vs. central	pooled	pooled	0.37 (R^2)
Monn et al. (1998)	Adults, Switzerland, eight regions in Swiss (four urban/suburban, two rural and two alpine regions), one-wk averaged, one measurement each mo (the first wk of the mo) for each subject, between Dec 1993 to Dec 1994	Personal vs. outdoor	pooled	pooled	0.33 (R^2)

TABLE AX3.13 (cont'd). THE ASSOCIATION BETWEEN PERSONAL EXPOSURES AND AMBIENT CONCENTRATIONS

Study	Study Design	Association Variable	Location	Season	r_p, r_s, or R^2
Levy et al. (1998a)	Adults, 18 cities across 15 countries, two-day averaged, one measurement for each person, all people were measured on the same winter day in February or March 1996	Personal vs. outdoor	urban	winter	0.57 (r_s)
Kodama et al. (2002)	Junior high school students and their family members, Tokyo, three-day averaged, samples were simultaneously collected on Feb 24-26, Jun 2-4, July 13-15, and Oct 14-16 in 1998 and Jan 26-28 in 1999	Personal vs. outdoor	urban	summer	0.24 (r_p)
		Personal vs. outdoor	urban	winter	0.08 (r_p)
Liard et al. (1999)	Adults and Children, Paris, 4-day averaged, three measurements for each person, during each measurement session, all subjects were measured at the same time during May/June 1996	Adults vs. central	urban	summer	0.41 (R^2)
		Children vs. central	urban	summer	0.17 (R^2)
Gauvin et al. (2001)	Children, three French metropolitan areas, 48-h averaged, one measurement for each child, all children in the same city were measured on the same day. The study occurred between April-June 1998 in Grenoble, May-June 1998 in Toulouse, and June-Oct 1998 in Paris.	Personal vs. central (Grenoble)	urban	pooled	0.01 (R^2)
		Personal vs. central (Toulouse)	urban	pooled	0.04 (R^2)
		Personal vs. central (Paris)	urban	pooled	0.02 (R^2)
Spengler et al. (1994)	Probability based population, Los Angeles Basin, 48-h averaged, one measurement per person in one of the eight sampling cycles (microenvironmental component of the study), from May 1987 to May 1988	Personal vs. outdoor	pooled	pooled	0.48 (R^2)
Kousa et al. (2001)	Probability based population, Helsinki, Basel, and Prague, 48-h averaged, one measurement per person, during 1996 and 1997	Personal vs. outdoor	urban	pooled	0.40 (R^2)

TABLE AX3.13 (cont'd). THE ASSOCIATION BETWEEN PERSONAL EXPOSURES AND AMBIENT CONCENTRATIONS

Study	Study Design	Association Variable	Location	Season	r_p, r_s, or R^2
Linaker et al. (2000)	Asthmatic children, Southampton, one-wk averaged, 13 mos for each child, until Dec 1995	Personal vs. outdoor (Overall measurements across children and time)	pooled, urban, no major indoor sources	pooled	Not significant
		Personal vs. outdoor (subject-wise)	By person	pooled	−0.77 to 0.68 and median −0.02 (r_p)
Lai et al. (2004)	Adults, Oxford, 48-h averaged, once per person, between Dec 1998 and Feb 2000	Personal vs. outdoor	urban	pooled	0.41 (r_p)
Kim et al. (2006)	Coronary artery adults, Toronto, 24-h averaged, one day a wk for 10 wks for each person, from Aug 1999 to Nov 2001	Personal vs. central (ambient)	urban	pooled	0.57 (r_s)
Sarnat et al. (2005)	Seniors and schoolchildren, Boston, 24-h averaged, 12 consecutive days in each of the 1 or 2 seasons, summer of 1999 and winter of 2000	Personal vs. central (subject wise)	urban	summer	−0.25 to 0.5 (r_s) with a median of 0.3*
				winter	−0.5 to 0.9 (r_s) with a median of 0.4*
Sarnat et al. (2006)	Seniors, Steubenville, 24-h averaged, the same two consecutive days each wk for 23 wks, summer and fall of 2000	Personal vs. central	urban	summer	0.14 (R^2)
				fall	0.43 (R^2)

* Values were estimated from figures in the original paper.

TABLE AX3.14. INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/ F_{inf}	Comments
Mosqueron et al. (2002)	48 h residential indoor, workplace, outdoor and personal exposure were measured for 62 Paris office workers using Ogawa badges from Dec 1999 to Sept 2000	Overall study seasons	Residential indoor vs. ambient and using gas cooking	Cooking	**0.26**	The overall R^2 is 0.14, and ambient NO_2 and indoor cooking account for 0.07 each.
			Office indoor vs. ambient and floor height	None	**0.56**	The overall R^2 is 0.24, partial R^2 for ambient and floor height were 0.18 and 0.06, respectively.
Lee et al. (1999)	The indoor and outdoor air quality of 14 public places with mechanical ventilation systems in Hong Kong; from Oct 1996 to March 1997; Teflon bags were used to collect indoor and outdoor NO and NO_2 during peak h	Overall study seasons	Indoor vs. outdoor	—	**0.59**	R^2 was 0.59. The slopes for NO and NO_x were 1.11 and 1.04 respectively.
			Indoor/outdoor ratio	—	0.53 – 1.03 (mean: 0.75)	0.83-2.68 for NO (mean: 0.99) 0.78-1.68 for NO_x (mean: 0.94)

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/ Finf	Comments
Monn et al. (1997)	During the SAPALDIA (Spain) study, 48–72 h indoor, outdoor, and personal NO_2 were measured by Palmes tubes between the winter of 1994 to the summer of 1995, and between May and July of 1996	Overall study seasons	Indoor/outdoor ratio	With gas-cooking	> 1.2	—
				Without gas cooking	**0.4-0.7**	—
Monn et al. (1997)	During the SAPALDIA (Spain) study, 48–72 h indoor, outdoor, and personal NO_2 were measured by Palmes tubes between the winter of 1994 to the summer of 1995, and between May and July of 1996	Overall study seasons	Indoor/outdoor ratio	With gas-cooking	> 1.2	—
				Without gas cooking	0.4-0.7	—
García-Algar et al. (2003)	Yanagisawa passive filter badges were used to measure indoor NO_2 concentrations for 7~15 days for 340 homes in Barcelona, Spain during 1996~1999. Outdoor NO_2 concentrations were obtained from the fixed monitoring stations by the method of CL.	Overall study seasons	Indoor/outdoor ratio		0.8-1.0	Including both homes with and without indoor sources.

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/F_{inf}	Comments
Lee et al. (1995)	Two-wk averaged indoor (kitchen, living room, and bedroom) and outdoor NO_2 were measured by Palmes tube for 517 homes from November 1984 to Oct 1986 in Boston area.	Summer	Indoor/outdoor ratio	Electric stove homes	0.81 (kitchen) 0.81 (living room) **0.77** (bedroom)	Homes with gas stove and gas stove with pilot light have an I/O ratio >1, but the values were not reported.
Lee et al. (2002)	Six-day integrated indoor and outdoor concentrations of NO_2 in two communities in Southern California were measured using Yanagisawa badges for 119 homes in April and May 1996.	Overall study seasons	Indoor/outdoor ratio ± SD	With gas range	2.27 ± 1.88	—
				Without gas range	1.22 ± 0.52	—
				With air conditioner	1.07 ± 0.26	—
				Without air conditioner	3.03 ± 2.01	—
Lee (1997)	Indoor and outdoor air quality at two staff quarters in Hong Kong were measured from January to Feb of 1996 by Chemical Luminescent method in two staff quarters in Hong Kong (TSTE, in a downtown area; and ST in a suburban area).	Overall study seasons	Indoor/outdoor ratio (Range)	Downtown area	0.78 (0.70-0.87) for NO_2 0.92 (0.77-1.10) for NO 0.86 (0.78-0.95) for NO_x	—
				Suburban area	0.97 (0.89-1.03) for NO_2 0.92 (0.77-3.14) for NO 0.86 (0.89-1.03) for NO_x	—

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/ F_{inf}	Comments
Garrett et al. (1999)	Four-day averaged indoor (bedroom, living room, and kitchen) and outdoor NO_2 was monitored using Yanagisawa passive samplers for 80 homes in the Latrobe Valley, Victoria, Australia, in March-April 1994, and Jan-Feb, 1995.	Overall study seasons	Indoor/outdoor ratio	No major indoor sources (major sources were gas stove, vented gas heater, and smoking)	0.8	The ratio increased to 1.3, to 1.8 and to 2.2 for homes with one, two, and three major indoor sources.
Zota et al. (2005)	Two-wk integrated NO_2 was measured in 77 homes within three Boston public housing developments (low-income, urban neighborhoods, where asthma prevalence is high), using Palmes tubes. Homes were sampled between June 2002 and May 2003 for 2-wk periods with up to three sampling sessions in each home.	Overall study seasons	Residential indoor vs. residential outdoor	—	1.21	—
Yang et al. (2004)	Daily indoor and outdoor NO_2 concentrations were measured for 30 consecutive days in 28 house in Brisbane (between April and May in 1999), and for 21 consecutive days in 37 houses in Seoul (between June and Aug in 2000) using Yanagisawa badges.	Overall study seasons	Residential indoor vs. residential outdoor	Brisbane with electric range	0.65 ± 0.18	R^2 was 0.70.
				Brisbane with gas range	0.56 ± 0.12	R^2 was 0.57.
				Seoul with gas range	0.58 ± 0.12	R^2 was 0.52.
			Indoor/outdoor ratio	Brisbane	0.82 ± 0.41	—
				Seoul	0.88 ± 0.32	—

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/F_{inf}	Comments
Chao (2001)	48-h averaged indoor and outdoor NO, and NO_2 were measured in ten non-smoking residential buildings using Ogawa passive samplers in the summer of 1997 in Hong Kong.	Overall study seasons	Indoor/outdoor ratio	—	0.79 ± 0.30 (range: 0.75 – 1.36) for NO_2 0.98 ± 0.19 (range: 0.29 – 1.25) for NO	—
Kulkarni et al. (2002)	48-h averaged indoor and outdoor NO_2 were measured using passive filter badge sampler in the winter (Feb 1996) and summer of 1996 (April) for 43 residence in Mumbai.	Overall study seasons	Residential indoor vs. residential outdoor	Homes using LPG Homes using Kerosene	0.92 0.73	R^2 was 0.80. R^2 was 0.40.
Monn et al. (1998)	One-wk averaged indoor, outdoor, and personal NO_2 were measured for more than 500 subjects between Dec 1993 to Dec 1994 for a SAPALDIA study subpopulation, once per home.	Overall study seasons	Residential indoor vs. residential outdoor	All homes Homes without smokers and gas-cooking	**0.47** **0.40**	R^2 was 0.37. R^2 was 0.33.
			Residential indoor vs. residential outdoor + gas cooking + smoking + ventilation	All homes	0.55	Overall R^2 was 0.58, but partial R^2 cannot be derived.
			Indoor/outdoor ratio	All homes	0.7-0.8	—

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio / F_{inf}	Comments
Levy et al. (1998a); Spengler et al. (1996)	48-h averaged indoor, outdoor, and personal exposures to NO_2 were measured in 18 cities in 15 countries around the world during a 2-day period in Feb or March 1996.	Overall study seasons	Indoor/outdoor ratio	Boston, US	0.6 ± 0.4	—
				Ottawa, Canada	0.5 ± 0.2	—
				Mexico City, Mexico	1.9 ± 1.0	—
				London, UK	0.6 ± 0.4	—
				Watford, UK	0.8 ± 0.4	—
				Geneva, Switzerland	0.8 ± 0.6	—
				Kjeller, Norway	0.7 ± 0.4	—
				Kuopio, Finland	0.5 ± 0.5	—
				Berlin, Germany	0.3 ± 0.2	—
				Erfurt, Germany	0.8 ± 0.7	—
				Homes without gas stove	0.7	—
				Homes with gas stove	1.2	—
				Homes without kerosene heater	0.85	—
				Homes with kerosene heater	2.27	—
				Homes without gas space heater	0.96	—
				Homes with gas space heater	1.93	—
				Homes without gas water heater	0.94	—
				Homes with gas water heater	1.07	—
				Homes without smokers present	0.92	—
				Homes with smokers present	1.16	—

TABLE AX3.14 (cont'd). INDOOR/OUTDOOR RATIO AND THE INDOOR VS. OUTDOOR REGRESSION SLOPE

Study	Description	Season	Regression Format or Ratio	Indoor Characteristics	Slope/Ratio/ F_{inf}	Comments
Spengler et al. (1994)	A Yanagisawa type of passive sample was used to measure the 48-h integrated indoor, outdoor and personal NO_2 levels from the May of 1987 to the May of 1988.	Overall study seasons	Residential indoor vs. residential outdoor	Gas range with pilot light	**0.49***	R^2 was 0.44.
				Gas range without pilot light	**0.4***	R^2 was 0.39.
				Electric stove	**0.4***	R^2 was 0.41.
Lai et al. (2004)	48-h averaged personal, indoor, outdoor and workplace NO_2 levels were measured by passive filter badges for 50 adults in Oxford between 1998 and 2000, once per person.	Overall study seasons	Indoor/outdoor ratio	All homes	0.9	—
				Smoking homes	1.5	—
				Non-smoking homes	1	—

Note: *Only data that are marked by underline and bold font can be considered as an infiltration factor.

TABLE AX3.15. NO_2 CONCENTRATIONS (PPB) IN DIFFERENT ROOMS

Study	Conditions	Outdoor	Kitchen	Living Room	Bedroom	Comments
Topp et al. (2004)	First visit	12.4	—	7.8	7.2	Indoor and outdoor NO_2 concentrations for 777 residential homes in five study areas were measured: Erfurt, Hamburg, Zerbst, Bitterfeld and Hettstedt during two visits (from June 1995 to May 1997, and from April 1996 to Sept 1998). In the study, one-week averaged NO_2 were measured by Palmes tube.
	Second visit	12.5	—	8.0	7.6	
Garrett et al. (1999)	No identified indoor sources	4.7	3.8	3.8	3.0	Garrett (1999) investigated the levels and sources of NO_2 in Australian homes. During the study, four-day averaged NO_2 was monitored using Yanagisawa passive samplers in 80 homes in the Latrobe Valley, Victoria in March-April 1994, and Jan-Feb 1995.
	Gas stove homes	4.7	8.0	6.7	6.3	
	Gas heater homes	4.7	7.3	6.9	5.0	
	Smoking homes	4.7	6.6	6.0	5.7	
	Homes with multiple sources	4.7	10.7	14.5	11.2	
Cotterill et al. (1997)	Gas Stove homes	20.9	35.6	17.3	11.5	Three consecutive two-week averaged outdoor, kitchen, living room, and bedroom NO_2 were measured using Palmes tubes in 40 houses in Huddersfield, UK in late 1994. Half the houses were located close to a busy main road and half on residential roads set back and parallel to the main road. The sample was split so that half had gas cookers and half had electric cookers. These subsets were split again so that half had double glazing and half had single glazed windows.
	Electric cooker homes	20.9	9.9	8.9	7.3	
	Gas cooker home with single glazing window	20.9	31.4	16.8	11.0	
	Gas cooker home with double glazing window	20.9	39.8	18.3	12.0	
Zota et al. (2005)	Overall	19	43	36	—	The indoor and outdoor NO_2 concentrations for low-income, urban neighborhoods were measured, where asthma prevalence is high. NO_2 was measured in 77 homes within three Boston public housing developments, using Palmes tubes (two-wk integrated sample) placed in the kitchen, living room, and outdoors. Air exchange rate for each home was also measured.
	Heating season	21	50	43	—	
	Non-heating season	17	33	26	—	

TABLE AX3.15 (cont'd). NO_2 CONCENTRATIONS (PPB) IN DIFFERENT ROOMS

Study	Conditions	Outdoor	Kitchen	Living Room	Bedroom	Comments
Gallelli et al. (2002)	Overall study	—	24.6	—	13.0	During the study, one-wk integrated indoor (kitchen and bedroom) and personal NO_2 were measured in Genoa, Italy, for 89 subjects with Palmes samplers. Study volunteers included students, workers, and housewives living in three areas of Genoa differing by street traffic and industrial plant location.
	With vent	—	18.1	—	—	
	Without vent	—	30.9	—	—	
Linaker et al. (1996)	Overall study	—	27.2	20.9	—	During the study, one-wk integrated personal, indoor (kitchen, living room), classroom, and playground NO_2 were measured using Palmes tubes for school children in Southampton.
Kodama et al. (2002)	Feb 1998	40, 31.3	81.8	73.5	55.2	The first number in outdoor column was the ambient concentration in the South Area; and the second number is the ambient concentration in the North Area. During the study, personal, indoor (kitchen, living room, bedroom and study room), and outdoor NO_2 were measured for 150 junior high school students with Yanagisawa badges in Tokyo. The investigation was conducted five times seasonally, 3 days each, from February 1998 to January 1999.
	June 1998	38, 28	33.2	28.8	24	
	July 1998	29, 26.7	24.8	21.9	17.4	
	Oct 1998	40, 35	23.5	24.7	18.2	
	Jan 1999	49, 50	70.9	65.8	50.7	
Chao and Law (2000)	Overall study	37.6	31.9	28.2	26.4	Personal and indoor exposures were monitored with passive sampler in Hong Kong for 60 subjects. Twelve of the subjects were selected to conduct more detailed study to examine the behavioral and microenvironmental effects on personal exposure to NO_2.

TABLE AX3.16. INDOOR AND OUTDOOR CONTRIBUTIONS TO INDOOR CONCENTRATIONS

Study	Condition	Slope	Intercept	Mean Indoor Concentration	Mean Outdoor Concentration	Percent Outdoor Contribution	Percent Indoor Contribution	Indoor Source Strength	Comments
Mosquero n et al. (2002)	Overall study	0.258	—	18.4	31.5	44.2	55.8	—	—
Yang et al. (2004)	Brisbane, electric range	0.65	0.8	10.3	—	92.4	7.6	3.5 ppb/h	—
	Brisbane, gas range	0.56	3.0	18.3	—	83.5	16.5	11.5 ppb/ h	—
	Seoul, gas range	0.58	4.8	33.4	40.4	85.7	14.3	23.4 ppb/ h	—
Monn et al. (1998)	Overall study	0.47	3.2	11.0	16.2	70.5	29.5	—	—
	Homes without smokers and gas cooking	0.40	3.2	6.8	16.2	53.1	46.9	—	Mean indoor was estimated based on the text description.

TABLE AX3.16 (cont'd). INDOOR AND OUTDOOR CONTRIBUTIONS TO INDOOR CONCENTRATIONS

Study	Condition	Slope	Intercept	Mean Indoor Concentration	Mean Outdoor Concentration	Percent Outdoor Contribution	Percent Indoor Contribution	Indoor Source Strength	Comments
Spengler et al. (1994)	Gas range with pilot light	0.49	—	30	37	60.4	39.6	—	Mean indoor and mean outdoor are estimated from Figure 2 in Spengler et al. (1994).
	Gas range without pilot light	0.4	—	22	33	60.0	40.0	—	Mean indoor and mean outdoor are estimated from Figure 2 in Spengler et al. (1994).
	Electric stove	0.4	—	17	33	77.6	22.4	—	Mean indoor and mean outdoor are estimated from Figure 2 in Spengler et al. (1994).
	Overall	0.49	8.64	27.2	38.3	68.2	31.8	—	—

TABLE AX3.17. THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Mosqueron et al. (2002)	Simultaneous personal, indoor, and in-office 48-h averaged NO_2 concentrations were measured with Ogawa badges for 62 people, and ambient concentrations were provided by local air monitoring network.	Overall study	0.07 (partial R^2)	—	—	Gas cooking interpreted another 7% of indoor NO_2 variation
Emenius et al. (2003)	Palmes tubes were used to measure indoor (in the main living room) and outdoor (outside the window of this room) NO_2 concentrations during a four-wk period (mean 28 days, range 26-31) in the first winter season following recruitment in the case-control study.	Without smoker and gas stove was not used	0.69 (r_p)	—	—	$p < 0.001$
		With gas stove and with smoker	0.13 (r_p)	—	—	$p = 0.43$
		With gas stove but without smoker	0.06 (r_p)	—	—	$p = 0.75$

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Lee et al. (1999)	Indoor and outdoor air quality of 14 public places with mechanical ventilation systems in Hong Kong were measured from Oct 1996 to March 1997. Traffic peak h NO, NO_2 was sampled using Teflon bags and then shipped back to the laboratory for further analysis.	Overall study	0.59 (R^2)	—	—	0.92 for NO and 0.92 for NO_x.
García-Algar et al. (2003)	Yanagisawa passive filter badges were used to measure indoor NO_2 concentrations for 7~15 days for 340 homes in Barcelona, Spain during 1996~1999. Outdoor NO_2 concentrations were obtained from the fixed monitoring stations by the method of CL.	Overall study	0.15 (r_p)	—	—	p = 0.007

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Lai et al. (2006)	The study was conducted between 1996 and 2000 in six EU cities: Athens, Basel, Helsinki, Milan, Oxford, and Prague. 48 h averaged indoor and outdoor NO_2 were collected each home using diffusion tubes for 302 homes.	Overall study	0.13 (partial R^2)	—	—	The overall R^2 for the multiple linear regression was 0.67
Lee et al. (2002)	Six-day integrated indoor and outdoor concentrations of NO_2 were measured in two communities in Southern California using Yanagisawa badges for 119 homes in April and May 1996.	Overall study	0.60 (r_p)	—	—	—
Mukala et al. (2000)	The one-week averaged indoor (day-care center), outdoor (outside day care center) and personal NO_2 for 162 children aged 3-6 years old nitrogen dioxide exposure were measured by Palmes tube in Helsinki, in 1991.	Spring	0.86 (r_p)	—	—	—
		Winter	0.54 (r_p)	—	—	—
		Spring (ambient vs. indoor)	0.45 (r_p)	—	—	—
		Winter (ambient vs. indoor)	0.36 (r_p)	—	—	—

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Garrett et al. (1999)	Four-day averaged NO_2 was monitored using Yanagisawa passive samplers in 80 homes in the Latrobe Valley, Victoria, Australia in March-April 1994, and Jan-Feb 1995.	Overall study	0.28 (R^2)	—	—	Log10 transformed data
Cotterill et al. (1997)	Three consecutive two-week averaged outdoor, kitchen, living room, and bedroom NO_2 were measured using Palme's tubes in 40 houses in Huddersfield, UK in late 1994. Half the houses were located close to a busy main road and half on residential roads set back and parallel to the main road. The sample was split so that half had gas cookers and half had electric cookers. These subsets were split again so that half had double glazing and half had single glazed windows.	Overall study	0.59 (r_p)	—	—	—

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Yang et al. (2004)	Daily indoor and outdoor NO_2 concentrations were measured for 30 consecutive days in 28 house in Brisbane (between April and May in 1999), and for 21 consecutive days in 37 houses in Seoul (between June and Aug in 2000) using Yanagisawa badges.	Brisbane, electric range house	0.70 (R^2)	—	—	—
		Brisbane, gas range house	0.57 (R^2)	—	—	—
		Seoul, gas range house	0.52 (R^2)	—	—	—
Lai et al. (2004)	During the study, 48-averaged personal, residential indoor, residential outdoor, and workplace indoor pollutants were measured for 50 adults between 1998 and 2000 in Oxford, once per person. NO_2 were measured using passive sampling badges.	Overall study	0.29 (r_p) (not significant)	0.47 (r_p) ($p < 0.01$)	-0.41 (r_p) ($p < 0.05$)	Data were log-transformed
Monn et al. (1998)	During the study, one-wk integrated indoor, outdoor and personal samples were collected for a subpopulation (n = 140) of SAPALDIA study using Pamles tube between Dec 1993 and Dec 1994 at eight study centers in Switzerland.	Overall study	0.37 (R^2)	0.51 (R^2)	0.33 (R^2)	—
		Homes without smoker and without gas-cooking	0.34 (R^2)	0.47 (R^2)	0.27 (R^2)	—

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Levy et al., (1998a)	48-h averaged indoor, outdoor and personal NO_2 were measured in 18 cities in 15 countries around the world with passive filter badges in Feb or March, 1996.	Overall study	—	0.75 (r_s)	0.57 (r_s)	—
Spengler et al. (1994)	Probability based population, Los Angeles Basin, 48-h averaged indoor, outdoor and personal NO_2 were measured (microenvironmental component of the study), from May 1987 to May 1988	Overall study	0.4 (R^2)	0.6 (R^2)	0.51 (R^2)	—
		Electric range	0.41 (R^2)	—	0.52 (R^2)	—
		Gas range without pilot light	0.39 (R^2)	—		—
		Gas range with pilot light	0.44 (R^2)	—	0.44 (R^2)	—
		With air conditioning	0.66 (r_p)	—	—	—
		Without air conditioning	0.75 (r_p)	—	—	—
		High ambient concentration	—	—	0.47 (R^2)	—
		Low ambient concentration	—	—	0.33 (R^2)	—

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Kousa et al. (2001)	The indoor, outdoor, and personal NO_2 relationship in three EXPOLIS centers (Basel, Helsinki, and Prague) were reported. During the study, 48-averaged indoor, outdoor, and personal NO_2 were measured with Palmes tubes during 1996-1997.	Overall study	0.44 (R^2)	0.53 (R^2)	0.37 (R^2)	Data were log-transformed
		Helsinki	—	0.45 (R^2)	0.40 (R^2)	Data were log-transformed
Linaker et al. (1996)	During the study, one-wk integrated personal, indoor (kitchen, living room), classroom and playground NO_2 were measured using Palmes tubes for 46 school children aged 9-11 in Southampton, UK.	Overall study	—	0.53-0.76 (r_p)	0.61-0.65 (r_p)	Data were log-transformed

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Alm et al. (1998)	During the study, weekly personal, indoor (day care center), outdoor (day care center), and ambient site NO_2 exposures of 246 children aged 3-6 yrs were measured with Palmes tubes during 13 wks in winter and spring in 1991 in Helsinki.	Overall study	—	0.88 (R^2)	0.86 (R^2)	0.37 (R^2) for personal vs. ambient
		Winter	—	—	0.04 (partial R^2)	p = 0.01; log transformed data
		Spring	—	—	0.50 (partial R^2)	p = 0.0001; log transformed data
		Winter downtown	0.44 (r_p)	0.32 (r_p)	0.46 (r_p)	Personal vs. indoor was not significant (day-care center, not residential indoor).
		Spring downtown	0.84 (r_p)	0.75 (r_p)	0.80 (r_p)	
		Winter suburban	0.22 (r_p)	0.04 (r_p)	0.49 (r_p)	Personal vs. indoor, and indoor vs. outdoor were not significant
		Spring suburban	0.46 (r_p)	0.75 (r_p)	0.82 (r_p)	—
		Downtown electric stove	—	0.67 (r_p)	0.55 (r_p)	—
		Downtown gas stove	—	0.50 (r_p)	0.59 (r_p)	—
		Downtown non-smoking	—	0.67 (r_p)	0.73 (r_p)	—
		Downtown smoking	—	0.47 (r_p)	0.51 (r_p)	—
		Suburban electric stove	—	0.55 (r_p)	0.63 (r_p)	—
		Suburban gas stove	—			—
		Suburban non-smoking	—	0.50 (r_p)	0.59 (r_p)	—
		Suburban smoking	—	0.48 (r_p)	0.46 (r_p)	—

TABLE AX3.17 (cont'd). THE ASSOCIATION BETWEEN INDOOR, OUTDOOR, AND PERSONAL NO_2

Study	Summary	Condition	Indoor vs. Outdoor	Personal vs. Indoor	Personal vs. Outdoor	Comments
Kodama et al. (2002)	During the study, personal, indoor (kitchen, living room, bedroom, and study room), and outdoor NO_2 were measured for 150 junior high school students with Yanagisawa badges in Tokyo. The investigation was conducted five times seasonally, 3 days each, from Feb 1998 to Jan 1999.	Summer	—	0.31 (r_p)	0.24 (r_p)	—
		Winter	—	0.57 (r_p)	0.08 (r_p)	—

TABLE AX3.18. INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Environmental conditions									
Singer et al. (2004)	Wind Direction	Upwind of freeway	20.5	—	—	—	—	—	—
		Downwind and close to freeway	26.5	—	—	—	—	—	—
		Downward and far from freeway	21	—	—	—	—	—	—
Zota et al. (2005)	Season	Heating	21	—	43	—	—	—	—
		Non-Heating	17	—	26	—	—	—	—
Sørensen et al. (2005)	Season	< 8C	14.6	—	8.9	—	11.4	—	—
		> 8C	7.8	—	6.6	—	9.2	—	—
Alm et al. (1998)	Season	Winter downtown smoker	—	—	—	—	13.5	—	—
		Spring downtown smoker	—	—	—	—	15.4	—	—
		Winter downtown nonsmoker	—	—	—	—	13.0	—	—
		Spring downtown nonsmoker	—	—	—	—	14.1	—	—
		Winter suburban smoker	—	—	—	—	11.2	—	—
		Spring suburban smoker	—	—	—	—	10.7	—	—
		Winter suburban nonsmoker	—	—	—	—	9.2	—	—
		Spring suburban nonsmoker	—	—	—	—	8.7	—	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Zota et al. (2005)	Heating season	—	—	3.87	—	17.3	—	—	—
Vukovich et al. (2000)	Day	Weekday		—	—	—	—	—	39% more than weekend
Lee (1997)	Day	Weekday	—	—	—	—	—	—	The effect of weekday/week-end is clear but the paper didn't give a value to cite
		Weekend	—	—	—	—	—	—	—
Dwelling conditions									
Levy et al. (1998a)	Window open	With	—	—	—	—	30	—	—
		Without	—	—	—	—	26.7	—	—
Cotterill et al. (1997)	Window	Single Glazing	—	—	9.4	—	—	—	—
		Double Glazing	—	—	9.4	—	—	—	—
		Single Glazing	—	—	11.0	—	—	—	Gas cooker homes
		Double Glazing	—	—	12.0	—	—	—	Gas cooker homes
Partti-Pellinen et al. (2000)	Type of Filtration	Mechanical filter	12.3	—	9.6	—	—	—	—
		Mechanical intake and mechanical filter	11.5	—	12.5	—	—	—	—
		Mechanical intake and mechanical and chemical filter	12.4	—	6.5	—	—	—	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Yanmanaka et al. (1984)	Surface type	—	—	—	—	—	—		Affect decay rate
Zota et al. (2005)	Occupancy	—	—	—	—	3.2	—	—	—
Levy et al. (1998a)	Occupancy	1	—	—	—	—	25.9	—	—
		2	—	—	—	—	30.8	—	—
Emenius et al. (2003)	Location	Urban	16.5	—	9.6	—	—	—	—
		Semi-urban	11.3	—	6.4	—	—	—	—
		Suburban	7.2	—	4.2	—	—	—	—
Cotterill et al. (1997)	Location	On Main Road	—	—	7.9	—	—	—	Electric cooker homes
		50-85m from Main Road	—	—	6.8	—	—	—	Electric cooker homes
Zota et al. (2005)	Location	—	—	-0.0093	—	—	—	—	—
Lee et al. (2004)	Location	Industrial	—	—	—	—	34.9	—	—
		Residential	—	—	—	—	27.8	—	—
Liard et al. (1999)	Location	Main Road	—	—	—	—	28.1	—	—
		Side Road	—	—	—	—	24.3	—	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Nakai et al. (1995)	Location	< 20 m	42.4	—	43.8	—	43.1	—	Recalculated based published data
		20-150 m	34.9	—	38.4	—	35.9	—	Recalculated based published data
		> 150 m	20.3	—	36.4	—	30.1	—	Recalculated based published data
Alm et al. (1998)	Location	Downtown smoker	—	—	—	—	14.6	—	—
		Suburban smoker	—	—	—	—	10.9	—	—
		Downtown nonsmoker	—	—	—	—	13.6	—	—
		Suburban nonsmoker	—	—	—	—	9.0	—	—
Lee et al. (1996)	House structure	Single DU	17	—	17	—	—	—	Winter
		Small multi-DU	23	—	28.9	—	—	—	Winter
		Large multi-DU	23.6	—	26.8	—	—	—	Winter
		Single DU	18.4	—	17.8	—	—	—	Fall
		Small multi-DU	25.1	—	30.2	—	—	—	Fall
		Large multi-DU	25.1	—	25.4	—	—	—	Fall
		Single DU	15.9	—	17.3	—	—	—	Summer
		Small multi-DU	23.7	—	27.8	—	—	—	Summer
		Large multi-DU	24.5	—	29.1	—	—	—	Summer

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Gallelli et al. (2002)	Heating system	Individual	—	—	13.7	—	—	—	Bedroom data
		Central	—	—	12.5	—	—	—	Bedroom data
	Frames	Metal	—	—	12.6	—	—	—	Bedroom data
		Wood	—	—	15.0	—	—	—	Bedroom data
Zota et al. (2005)	Floor level	—	—	2	—	—	—	—	—
Mosqueron et al. (2002)	Floor level	—	—	—	—	−1.78	—	—	—
Liard et al. (1999)	Extractor fan over cooker	Without	—	—	—	—	27.5	—	—
		With	—	—	—	—	24.8	—	—
Gallelli et al. (2002)	Chimney	With vent	—	—	18.1	—	—	—	Kitchen data
		Without vent	—	—	30.9	—	—	—	Kitchen data
Yang et al. (2004)	Attached garage	With	—	—	17.3	—	—	—	—
		Without	—	—	11.4	—	—	—	—
Garrett et al. (1999)	Age of house	—	—	—	—	0.5	—	—	—
Indoor sources									
Zota et al. (2005)	Supplemental Heating with stove	—	—	—	—	7.84	—	—	—
Lai et al. (2004)	Smoking	Smoking	—	—	10.9	—	10.8	—	—
		Nonsmoking	—	—	11.5	—	14.1	—	—
Levy et al. (1998a)	Smokers present	With	—	—	—	—	34.8	—	—
		Without	—	—	—	—	26.8	—	—
Belanger et al. (2006)	Ranges	Electric	—	—	8.6	—	—	—	—
		Gas	—	—	25.9	—	—	—	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Cotterill et al. (1997)	Ranges	Gas	—	—	35.6	—	—	—	Kitchen
		Electric	—	—	9.9	—	—	—	Kitchen
		Gas	—	—	11.5	—	—	—	Bedroom
		Electric	—	—	7.3	—	—	—	Bedroom
Yang et al. (2004)	Ranges	Gas	—	—	18.3	—	—	—	—
		Not Gas	—	—	10.3	—	—	—	—
Schwab et al. (1994)	Ranges	Gas with pilot light	—	—	20.3	—	—	—	Summer 1998 data
		Gas without pilot light	—	—	11.7	—	—	—	Summer 1998 data
		Electric	—	—	8	—	—	—	Summer 1998 data
Monn et al. (1998)	Ranges	Gas Geneva	—	—	20.9	—	23.6	—	—
		Electric Geneva	—	—	16.8	—	19.9	—	—
		Gas Basle	—	—	15.2	—	18.3	—	—
		Electric Basle	—	—	12.6	—	16.2	—	—
		Gas Lugano	—	—	18.8	—	20.9	—	—
		Electric Lugano	—	—	15.7	—	18.3	—	—
Spengler et al. (1994)	Ranges		—	—	—	—	—	—	Gas with pilot was 15 ppb higher than electric; gas without pilot was 4 ppb higher than electric
Alm et al. (1998)	Ranges	Electric smoker	—	—	—	—	13.0	—	—
Raaschou-Nielsen et al. (1997)	Near fire		—	—	—	—	—	0.052	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Kawamoto et al. (1997)	Heating time	Oil fan heater	—	—	—	—	—	2.59	—
		Kerosene heater	—	—	—	—	—	1.17	—
		Clean heater	—	—	—	—	—	—	—
Lee et al. (2004)	Heating fuel	Coal briquette	—	—	—	—	22.2	—	—
		Petroleum	—	—	—	—	33.1	—	—
Liard et al. (1999)	Heating appliance	Gas	—	—	—	—	27.9	—	—
		Other	—	—	—	—	25.2	—	—
Kodama et al. (2002)	Heater	Kerosene heater	—	—	152.6	—	—	—	Sourth area, Feb 1998
		Gas stove	—	—	77.5	—	—	—	Sourth area, Feb 1998
		Electric heater	—	—	30.8	—	—	—	Sourth area, Feb 1998
Yang et al. (2004)	Gas water heater	With	—	—	18.1	—	—	—	—
		Without	—	—	11.9	—	—	—	—
Levy et al. (1998a)	Gas water heater	With	—	—	—	—	30.5	—	—
		Without	—	—	—	—	28.2	—	—
		With	—	—	—	—	36.4	—	—
		Without	—	—	—	—	28.5	—	—
	Gas range	With	—	—	—	—	34.8	—	—
		Without	—	—	—	—	20.5	—	—
Monn et al. (1997)	Gas cooking	With	—	—	—	—	—	—	I/O > 1.2
		Without	—	—	—	—	—	—	I/O ~ 0.4 – 0.7
Mosqueron et al. (2002)	Gas cooking		—	—	—	0.068	—	—	—
Raaschou-Nielsen et al. (1997)	Gas appliances at home		—	—	—	—	—	0.202	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Garrett et al. (1999)	Gas and smoking	None	—	—	3.0	—	—	—	I/O ratio increase from 0.8 to 1.3 to 1.8 to 2.2 in houses with no, one, two, or three major indoors sources
		Gas stove	—	—	6.3	—	—	—	—
		Gas heater	—	—	5.0	—	—	—	—
		Smoking	—	—	5.7	—	—	—	—
		Multiple	—	—	11.2	—	—	—	—
Dutton et al. (2001)	Fireplace setting	Low	—	—	90	—	—	—	—
		Middle	—	—	350	—	—	—	—
		High	—	—	360	—	—	—	—
Sørensen et al. (2005)	Exposure to burning candle	—	—	—	—	—	—	0.031	—
Liard et al. (1999)	Exposure to ETS	With	—	—	—	—	25.1	—	—
		Without	—	—	—	—	26.3	—	—
Raaschou-Nielsen et al. (1997)	Exposure to ETS		—	—	—	—	—	0.056	—
Lee et al. (2004)	Cooking fuel	Petroleum	—	—	—	—	26.1	—	—
		Gas	—	—	—	—	33.1	—	—
		Coal briquette	—	—	—	—	20.6	—	—
Liard et al. (1999)	Cooking appliance	Gas	—	—	—	—	25.8	—	—
		Electric	—	—	—	—	25.5	—	—

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Dennekamp et al. (2001)	Cooking	1 ring	—	—	437	—	—	—	The max 5 min concentrations
		2 rings	—	—	310	—	—	—	The max 5 min concentrations
		3 rings	—	—	584	—	—	—	The max 5 min concentrations
		4 rings	—	—	996	—	—	—	The max 5 min concentrations
		Boil water	—	—	184	—	—	—	The max 5 min concentrations
		Stir fry	—	—	92	—	—	—	The max 5 min concentrations
		Fry bacon	—	—	104	—	—	—	The max 5 min concentrations
		Bake cake	—	—	230	—	—	—	The max 5 min concentrations
		Roast meat	—	—	296	—	—	—	The max 5 min concentrations
		Bake potatoes	—	—	373	—	—	—	The max 5 min concentrations

TABLE AX3.18 (cont'd). INDOOR, OUTDOOR, AND PERSONAL NO_2 LEVELS STRATIFIED BY EXPOSURE INDICATORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Name	Factor Levels	Ambient NO_2 Level	Ambient Slope	Indoor NO_2 Level	Indoor Slope	Personal NO_2 Level	Personal Slope	Comments
Personal activities									
Levy et al. (1998a)	Commute	Commuting less than 1 h	—	—	—	—	29.9	—	—
		Without commuting	—	—	—	—	27.9	—	—
Chao and Law (2000)	Commute	< 1 h	—	—	—	—	21.7	—	—
		1-2 h	—	—	—	—	24.7	—	—
		2-3 h	—	—	—	—	24.6	—	—
		3-4 h	—	—	—	—	20.1	—	—
		4-6 h	—	—	—	—	27.9	—	—
	Cooking to stay home h ratio	—	—	—	—	—	—	55.4	—
Kawamoto et al. (1997)	Cooking time	—	—	—	—	—	—	1.61	—

TABLE AX3.19. PERSONAL NO_2 LEVELS STRATIFIED BY DEMOGRAPHIC AND SOCIOECONOMIC FACTORS (CONCENTRATIONS ARE IN PPB AND SLOPES ARE DIMENSIONLESS)

References	Factor Type	Factor Name	Factor levels	Personal NO_2 Level	Personal Slope
Rotko et al. (2001)	Demography	Age	25-34	13.1	
Rotko et al. (2001)	Demography	Age	35-55	13.1	
Raaschou-Nielsen (1997)	Demography	Age			0.056
Lee et al., (2004)	Demography	Gender	Female	33	
Lee et al., (2004)	Demography	Gender	Male	29	
Rotko et al. (2001)	Demography	Gender	Female	12.9	
Rotko et al. (2001)	Demography	Gender	Male	13.4	
Raaschou-Nielsen (1997)	Demography	Gender			0.267
Rotko et al. (2001)	Socioeconomic	Education years	<14 years	13.8	
Rotko et al. (2001)	Socioeconomic	Education years	≥14 years	12.8	
Rotko et al. (2001)	Socioeconomic	Employment	Employed	13.3	
Rotko et al. (2001)	Socioeconomic	Employment	Not employed	11.5	
Rotko et al. (2001)	Socioeconomic	Occupational status	Non white collar	13.4	
Rotko et al. (2001)	Socioeconomic	Occupational status	White collar	13.0	
Algar et al. (2004)	Socioeconomic	Employment	Managerial, technical and professional (Barcelona)	12.2	
Algar et al. (2004)	Socioeconomic	Employment	Skilled (manual and non-manual) (Barcelona)	12.3	
Algar et al. (2004)	Socioeconomic	Employment	Unskilled and partly skilled (Barcelona)	12.1	

TABLE AX3.20. CORRELATIONS (PEARSON CORRELATION COEFFICIENT) BETWEEN AMBIENT NO_2 AND AMBIENT COPOLLUTANTS

Study (ambient)	Location	$PM_{2.5}$	CO	O_3	SO_2
This CD	Los Angeles	0.49 (u[3]), 0.56 (s)	0.59 (u), 0.64 (s)	−0.29 (u), −0.11 (s)	
This CD	Riverside, CA		0.43 (u), 0.41 (s), 0.15 (r)	0.045 (u), 0.10 (s), −0.31 (r)	
This CD	Chicago	0.49 (s)	0.53 (u), 0.46 (s)	−0.20 (u)	
This CD	New York City	0.58 (u)	0.46 (u)	−0.06 (u)	
Kim et al. (2006)	Toronto	0.44	0.72		
Sarnat et al. (2006)	Steubenville, OH (autumn)	0.78 (0.70 for sulfate, 0.82 for EC)			
Sarnat et al. (2006)	Steubenville, OH (summer)	0.00 (0.1 for sulfate, 0.24 for EC)			
Connell et al. (2005)	Steubenville, OH	0.50			
Kim et al. (2005)	St. Louis (RAPS)		0.64[1]		
Sarnat et al. (2001)[4]	Baltimore, MD (summer)	0.37	0.75	0.02 not significant	
Sarnat et al. (2001)	Baltimore, MD (winter)	0.75	0.76	−0.71	-0.17
Hochadel et al. (2006)	Ruhr area, Germany	0.41, (0.93 for EC[2])			
Hazenkamp-von Arx et al. (2004)	21 European cities	0.75			
Cyrys et al. (2003)	Erfurt, Germany	0.50	0.74		
Mosqueron et al. (2002)	Paris	0.69			
Rojas-Bracho et al. (2002)	Santiago, Chile	0.77			

[1]Value with respect to NO_x.
[2]Inferred based on EC as dominant contributor to $PM_{2.5}$ absorbance.
[3]u: urban; s: suburban; and r: rural
[4]Spearman correlation coefficient was reported

TABLE AX3.21. CORRELATIONS (PEARSON CORRELATION COEFFICIENT) BETWEEN PERSONAL NO_2 AND PERSONAL COPOLLUTANTS

Study	Location	$PM_{2.5}$	CO	VOCs	HONO
Kim et al. (2006)	Toronto	0.41	0.12		
Modig et al. (2004)	Umea			0.06 for 1,3-butadiene; and 0.10 for benzene	
Mosqueron et al. (2002)	Paris	0.12 but not significant			
Jarvis et al. (2005)	21 European cities				0.77 for indoor NO_2 and indoor HONO
Lee et al. (2002)					0.51 for indoor NO_2 and indoor HONO
Lai et al. (2004)	Oxford	-0.1	0.3	−0.11 for TVOCs	

TABLE AX3.22. CORRELATIONS (PEARSON CORRELATION COEFFICIENT) BETWEEN PERSONAL NO_2 AND AMBIENT COPOLLUTANTS

Study	Location	$PM_{2.5}$	Sulfate	EC	PM_{10}	CO
Sarnat et al. (2006)	Steubenville / Fall	0.46	0.35	0.57		
Sarnat et al. (2006)	Steubenville / Summer	0.00	0.1 not significant	0.17		
Kim et al. (2006)	Toronto	0.30				0.20
Rojas-Bracho et al. (2002)	Santiago	0.65			0.39	

TABLE AX3.23. CORRELATIONS (PEARSON CORRELATION COEFFICIENT) BETWEEN AMBIENT NO_2 AND PERSONAL COPOLLUTANTS

Study	Location	$PM_{2.5}$	Sulfate	EC	Ultrafine-particle
Sarnat et al. (2006)	Steubenville / Fall	0.71	0.52	0.70	
Sarnat et al. (2006)	Steubenville / Summer	0.00	0.1 not significant	0.26	
Vinzents et al. (2005)	Copenhagen				0.49 (R^2) explained by ambient NO_2 and ambient temperature

TABLE AX3.24. THE ESSENTIAL ATTRIBUTES OF THE PNEM, HAPEM, APEX, SHEDS, AND MENTOR-1A

	pNEM	HAPEM	APEX	SHEDS	MENTOR-1A
Exposure Estimate	Hourly averaged	Annual averaged	Hourly averaged	Activity event based	Activity event based
Characterization of the High-End Exposures	Yes	No	Yes	Yes	Yes
Typical Spatial Scale/Resolution	Urban areas/Census tract level	Ranging from urban to national/ Census tract level	Urban area/Census tract level	Urban areas/Census tract level	Multiscale/ Census tract level
Temporal Scale/Resolution	A yr/one h	A yr/one h	A yr/one h	A yr/event based	A yr/activity event based time step
Population Activity Patterns Assembly	Top-down approach	Top-down approach	Bottom-up "person-oriented" approach	Bottom-up "person-oriented" approach	Bottom-up "person-oriented" approach
Microenvironment Concentration Estimation	Non-steady-state and steady-state mass balance equations (hard-coded)	Linear relationship method (hard-coded)	Non-steady-state mass balance and linear regression (flexibility of selecting algorithms)	Steady-state mass balance equation (residential) and linear regression (non-residential) (hard-coded)	Non-steady-state mass balance equation with indoor air chemistry module or regression methods (flexibility of selecting algorithms)
Microenviron-mental (ME) Factors	Random samples from probability distributions	Random samples from probability distributions	Random samples from probability distributions	Random samples from probability distributions	Random samples from probability distributions
Specification of Indoor Source Emissions	Yes (gas-stove, tobacco smoking)	Available; set to zero in HAPEM6	Yes (multiple sources defined by the user)	Yes (gas-stove, tobacco smoking, other sources)	Yes (multiple sources defined by the user)
Commuting Patterns	Yes	Yes	Yes	Yes	Yes
Exposure Routes	Inhalation	Inhalation	Inhalation	Inhalation	Multiple (optional)
Potential Dose Calculation	Yes	No	Yes	Yes	Yes
Physiologically Based Dose	No	No	No	Yes	Yes
Variability/ Uncertainty	Yes	No	Yes	Yes	Yes (Various "Tools")

AX3.9 REFERENCES

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AX4. CHAPTER 4 ANNEX – TOXICOLOGICAL EFFECTS OF NITROGEN DIOXIDE AND RELATED OXIDES OF NITROGEN

Effects of Nitrogen Dioxide on Antioxidant and Antioxidant Metabolism

Nitrogen dioxide is an oxidant and lipid peroxidation is believed to be a major molecular event responsible for its toxicity. As a result, there has been considerable attention paid to the effect of NO_2 on the antioxidant defense system in the epithelial lining fluid and in pulmonary cells. Repeated exposures to NO_2 at concentrations ranging from 75 to 62,040 $\mu g/m^3$ (0.04 to 33 ppm) have revealed effects on low molecular weight antioxidants such as glutathione, vitamin E, and vitamin C, as well as some enzymes involved in cell oxidant homeostasis.

A number of studies have investigated the hypothesis, originally proposed by Menzel (1970), that antioxidants might protect the lung from NO_2 damage by inhibiting lipid peroxidation (see Table AX4.1). Changes in the activity of enzymes in the lungs of NO_2-exposed animals that regulate levels of glutathione (GSH) have been reported at relatively low exposure concentrations. Sagai et al. (1984) studied the effects of prolonged (9 and 18 months) exposure to 75, 752, and 7520 $\mu g/m^3$ (0.04, 0.4, and 4.0 ppm) NO_2 on rats. After either exposure duration, non-protein sulfhydryl levels were increased at 752 $\mu g/m^3$ or greater, and exposure to 7520 $\mu g/m^3$ (4.0 ppm) decreased the activity of GSH peroxidase but increased glucose-6-phosphate dehydrogenase activity. Glutathione peroxidase activity was also decreased in rats exposed to 752 $\mu g/m^3$ NO_2 for 18 months. Three GSH S-transferases were also studied, two of which (aryl *S*-transferase and aralkyl *S*-transferase) exhibited decreased activities after 18 months of exposure to 752 $\mu g/m^3$ or greater NO_2. No effects were observed on the activities of 6-phosphogluconate dehydrogenase, superoxide dismutase, or disulfide reductase. Effects followed a concentration- and exposure-duration response function. The decreases in glutathione-related enzyme activities were inversely related to the apparent formation of lipid peroxides (see lipid peroxidation subsection). Shorter exposures (4 months) to NO_2 between 752 and 7520 $\mu g/m^3$ (0.4 and 4.0 ppm) also caused concentration- and duration-dependent effects on antioxidant enzyme activities (Ichinose and Sagai, 1982). For example, glucose-6-phosphate dehydrogenase increased, reaching a peak at 1 month, and then decreased

towards the control value. Briefer (2-week) exposures to 752 μg/m^3 (0.4 ppm) NO_2 caused no such effects in rats or guinea pigs (Ichinose and Sagai, 1989).

The activities of GSH reductase and glucose-6-phosphate dehydrogenase were significantly increased during exposure to11,700 μg/m^3 (6.2 ppm) NO_2 for 4 days; GSH peroxidase activity was not affected (Chow et. al., 1974). The possible role of edema and cellular inflammation in these findings was not examined. Since NO_2 had no significant effect on lung GSH peroxidase activity in this study, but did significantly increase the activities of GSH reductase and glucose-6-phosphate dehydrogenase, the authors concluded that NO_2 attacks mainly GSH and NADPH.

Newer studies also identified effects on glutathione. Changes in glutathione status in the blood and lung (bronchoalveolar lavage (BAL) fluid) occurred in rats exposed to 9400 μg/m^3 (5 ppm) and 18,800 μg/m^3 (10 ppm) NO_2 continuously for 24 h, but not for 7 days (Pagani et al., 1994). Total glutathione - total of reduced (GSH) and oxidized (GSSG) form - was significantly increased in blood but not in BAL fluid; however, GSSG was elevated in BAL fluid only. A decreased GSH/GSSG ratio was observed in the blood and BAL fluid, but not in lung type II cells, in rats continuously exposed to 18,800 μg/m^3 (10 ppm) NO_2 for 3 or 20 days (Hochscheid et al., 2005). Interestingly, lipid peroxidation was decreased in type II cells at 3 days, but was similar to controls at 20 days. Gene expression, as measured by mRNA levels of the enzymes involved in the biosynthesis of glutathione – gamma-glutamylcysteine synthetase (γGCS) and glutathione synthetase (GS), was decreased at both time points, but gamma-glutamyltranspeptidase (γGT) mRNA expression was increased. No GSH peroxidase activity (important for hydroperoxide reduction of complex lipids) was detected at 3 days, and was barely detected at 20 days.

Malnutrition of animals can drastically affect their response to toxicants, including NO_2. Experimental interest in this area has mainly focused on dietary lipids, vitamin E and other lipid-soluble antioxidants, and vitamin C and other water-soluble antioxidants. Ayaz and Csallany (1978) exposed vitamin E-deficient and vitamin E-supplemented (30 or 300 mg/kg opf diet) weanling mice continuously for 17 months to 940 or 1880 μg/m^3 (0.5 or 1.0 ppm) NO_2 and assayed blood, lung, and liver tissues for GSH peroxidase activity. Exposure to 1880 μg/m^3 (1.0 ppm) NO_2 alone or combined with vitamin E deficiency decreased the enzyme activity in the blood and lungs. Neither vitamin E deficiency nor NO_2 exposure affected liver GSH

peroxidase activity. However, in vitamin E-supplemented mice, GSH peroxidase activity increased at 940 μg/m^3 (0.5 ppm) and 1880 μg/m^3 (1.0 ppm) NO_2.

Lipid Metabolism and Content of the Lung

Lipid peroxidation is an important mechanism of cell damage arising from changes in cell membrane structure and function. The ability of NO_2 exposure to induce lipid peroxidation in the respiratory tract has been well demonstrated in available studies as measured by increased ethane exhalation in the breath, as thiobarbituric acid (TBA) reactive substances in tissues, and as the content of conjugated dienes in tissue homogenates.

A number of studies have investigated the effects of NO_2 exposure on lipid metabolism and content of the lung. Lipid peroxidation induced by NO_2 exposure has been detected at exposure concentrations as low as 75 μg/m^3 (0.04 ppm). Increased ethane exhalation was observed in rats exposed to 75 or 225 μg/m^3 (0.04 or 0.12 ppm) after 9 and 18 months of exposure (Sagai et al., 1984). Exposure to 752 μg/m^3 (0.4 ppm) NO_2 for 9 months or longer and to 7520 μg/m^3 (4.0 ppm) for 6 months resulted in increased TBA reactants (Ichinose et al., 1983). NO_2 exposures for shorter durations also increased lipid peroxidation in rats. For example, NO_2 concentrations of 2256 μg/m^3 (1.2 ppm) or greater for 1 week (Ichinose and Sagai, 1982; Ichinose et al., 1983) increased ethane exhalation in rats, while exposure of pregnant rats to 1000 μg/m^3 or 10,000 μg/m^3 (0.53 or 5.3 ppm) NO_2 for 5 h/day for 21 days rats resulted in increases in lung lipid peroxidation products (Balabaeva and Tabakova, 1985). These results indicate at least some degree of duration-dependence in the formation of lipid peroxidation, with lower effect thresholds identified with longer durations of exposure.

Lipid peroxidation results in altered phospholipid composition, which in turn may affect membrane fluidity and thus, membrane function. Significant depression of lipid content and total content of saturated fatty acids such as phosphatidyl-ethanolamine, lecithin (phosphatidylcholine), phosphatidylinositol, and phosphatidylserine were found in rats exposed to 5450 μg/m^3 (2.9 ppm) NO_2 for 24 h/day, 5 days/week for 9 months (Arner and Rhoades, 1973). Exposure of rabbits to 1880 μg/m^3 (1.0 ppm) NO_2 for 2 weeks also caused depression of lecithin synthesis after one week of exposure (Seto et al., 1975), while exposure of rats to 10,300 μg/m^3 (5.5 ppm) NO_2 for 3 h/day for 7 or 14 days elicited only few changes in lipid metabolism (Yokoyama et al., 1980). In beagle dogs, the amount of unsaturated fatty acids in the phospholipids from the lungs was increased after exposure to concentrations ranging from

9400 to 30,080 μg/m^3 (5 to 16 ppm), but not to 5640 μg/m^3 (3 ppm) (Dowell et al., 1971). Exposure of either mice or guinea pigs to an NO_2 level of 750 μg/m^3 (0.4 ppm) for a week resulted in a decreased concentration of phosphatidylethanolamine and a relative increase in the phosphatidylcholine concentration (Sagai et al., 1987). Concentration- and exposure duration-dependent increases were reported in phospholipid components in BAL fluid, when rats were exposed to 10 ppm NO_2 continuously for 1 day or 3 days (Müller et. al., 1994).

Functional studies conducted on surfactant phospholipid extracts indicated that NO_2 exposures of 5 ppm or greater, but not to 0.8 ppm, directly impaired surface tension, although the structure of the surfactant protein A (SP-A) was not altered by NO_2 exposure. Changes in the phospholipid composition of membranes may result in disruption of the cell membrane barrier. Müller et al. (2003) found that uptake of liposomes by type II lung cells occurred more easily from animals exposed to 10 ppm NO_2 for 3 to 28 days, possibly as a result of increased demand of phosphatidylcholine during lung injury.

Lipid peroxidation can also activate phospholipases. Activation of phospholipase A1 in cultured endothelial cells occurred at NO_2 concentration of 9400 μg/m^3 (5 ppm) after 40 h of exposure and was speculated to depend on a specific NO_2-induced increase in phosphatidyl serine in the plasma membranes (Sekharam et al., 1991).

One function of phospholipases is the release of arachidonic acid (AA), which serves as a mediator of inflammatory response. NO_2 exposure affects the release and metabolism of arachidonic acid both in vivo and in vitro. The products of arachidonic acid metabolism, such as prostaglandins, prostacyclin, thromboxanes, and leukotrienes play an important role (such as recruitment of neutrophils to sights of local irritation) in modulating inflammatory response. Schlesinger et al. (1990) reported elevated concentrations of thromboxane B2 (TxB_2) following NO_2 exposures of 1880 μg/m^3 (1.0 ppm) for 2 h, depressed concentrations at 5640 μg/m^3 (3.0 ppm), and significant depression 24 h postexposure at 18,880 μg/m^3 (10 ppm) NO_2. The same investigators also reported depressed level of 6-keto-prostaglandin F1α at 1880 μg/m^3 (1.0 ppm) NO_2, but exposure to NO_2 did not affect prostaglandins E2 and F2 and leukotriene B4 (LTB_4) levels.

Changes in activation of arachidonate metabolism were also reported in rat alveolar macrophages (AMs) when these animals were exposed to 940 μg/m^3 (0.5 ppm) NO_2 for 0.5, 1, 5, and 10 days (Robison et al., 1993). Unstimulated AM synthesis of LTB_4 was depressed after

0.5 days and again after 5 days of exposure to NO_2. Alveolar macrophage production of TxB_2, LTB_4, and 5-hydroxyeicosatetraenoic acid (5-HETE) in response to stimulation with the calcium ionophore, A23187, was depressed after 0.5 days of exposure and recovered to air-control values with longer exposure periods. 5-HETE levels were increased after 10 days of exposure. However, AM production of LTB_4 in response to zymosan-activated rat serum was depressed only after 5 days of exposure.

The effects of NO_2 on structural proteins of the lungs have been of concern because elastic recoil is lost after exposure. Collagen synthesis rates are increased in rats exposed to NO_2 concentrations as low as 9400 $\mu g/m^3$ (5.0 ppm) NO_2. It has been assumed that increased collagen synthesis reflect increases in total lung collagen which, if sufficient, could result in pulmonary fibrosis after longer periods of exposure. Such correlation has yet to be confirmed by in vivo studies involving NO_2 exposure.

Alterations in xenobiotic metabolism pathways following NO_2 exposure are also summarized in Table AX4.2, in addition to changes in phase I enzymes (such as cytochrome P450s) and phase II enzymes (GST as described earlier). While these changes are not necessarily toxic manifestations of NO_2 per se, such changes may impact the metabolism and toxicity of other chemicals. Glycolytic pathways are also apparently affected. For example, glycolytic metabolism was increased by NO_2 exposure, apparently due to a concurrent increase in type II cells (Mochitate et al., 1985).

Emphysema Following Nitrogen Dioxide Exposure

Emphysema as a result of chronic exposure to NO_2 has been reported in animal studies. The definition of emphysema has changed since the time that some of the studies have been published; thus, it is important to compare the findings of the studies with the current definition of emphysema. U.S. Environmental Protection Agency (1993) evaluated the animal studies reporting emphysema from chronic exposure to NO_2 based upon the most recent definition of emphysema from the report of the National Heart, Lung and Blood Institute (NHLBI), Division of Lung Diseases Workshop (Snider et al., 1985); see U.S. Environmental Protection Agency (1993) for the definitions of emphysema. Because the focus of this document is extrapolation of NO_2 exposures to potential hazards for humans, only those studies showing emphysema of the type seen in human lungs will be discussed.

Emphysema was reported by Haydon et al. (1967) in rabbits exposed continuously (presumably 24 h/day) for 3 to 4 months to 15,000 or 22,600 $\mu g/m^3$ (8.0 or 12.0 ppm) NO_2. The investigators reported enlarged lungs that failed to collapse when the thorax was opened. When the lungs were fixed in an expanded state via the trachea using formaldehyde, there was evidence of enlarged airspaces with destructive changes in alveolar walls. Although no stereology was performed, the changes observed appear to be emphysema of the type seen in human lungs.

WHO (1997) has also reported a study by Freeman et al. (1972) in which rats were exposed to 37,600 $\mu g/m^3$ (20.0 ppm) NO_2, which was reduced during the exposure to 28,200 $\mu g/m^3$ (15.0 ppm) or to 18,800 $\mu g/m^3$ (10.0 ppm), for varying periods up to 33 months. The lungs were fixed via the trachea, and morphometric analysis of the lung and alveolar size indicated an enlargement of alveolar, reduction in alveolar surface, and alveolar destruction. Although the investigators concluded that their study demonstrated emphysema in their NO_2-exposed rats, WHO (1997) noted that it was not entirely clear whether the experimental groups or only the group exposed to 28,200 $\mu g/m^3$ (15.0 ppm) had emphysema.

Although many of the papers reviewed (U.S. Environmental Protection Agency, 1993) reported finding emphysema, some of these studies were reported according to previous, different criteria; some reports did not fully describe the methods used; and/or the results obtained were not in sufficient detail to allow independent confirmation of the presence of emphysema. For example, Hyde et al. (1978) reported no emphysema in beagle dogs exposed 16 h daily for 68 months to 1200 $\mu g/m^3$ (0.64 ppm) NO_2 with 310 $\mu g/m^3$ (0.25 ppm) NO or to 263 $\mu g/m^3$ (0.14 ppm) NO_2 with 2050 $\mu g/m^3$ (1.67 ppm) NO. The dogs then breathed clean air during a 32- to 36-month post-exposure period. The right lungs were fixed via the trachea at 30-cm fixative pressure in a distended state. Semiautomated image analysis was used for morphometry of air spaces. The dogs exposed to 1200 $\mu g/m^3$ NO_2 with 310 $\mu g/m^3$ NO had significantly larger lungs with enlarged air spaces and evidence of destruction of alveolar walls. These effects were not observed in dogs exposed to 270 $\mu g/m^3$ NO_2 with 2050 $\mu g/m^3$ NO, implying a significant role of the NO_2 in the production of the lesions. The lesions in the dogs exposed to the higher NO_2 concentration meet the criteria of the 1985 NHLBI workshop for emphysema of the type seen in human lungs.

Nitrates (NO_3^-)

Busch et al. (1986) exposed rats and guinea pigs with either normal lungs or elastase-induced emphysema to ammonium nitrate aerosols at 1 mg/m^3 for 6 h/day, 5 days/week for 4 weeks. Using light and electron microscopy, the investigators concluded that there were no significant effects of exposure on lung structure.

AX4.2 DOSIMETRY OF INHALED NITROGEN OXIDES

This section provides an overview of NO_2 dosimetry and updates information provided in the 1993 AQCD for Oxides of Nitrogen. Dosimetry of NO_2 refers to the measurement or estimation of the amount of NO_2 or its reaction products reaching and persisting at specific sites in the respiratory tract following an exposure. Nitrogen dioxide, classified as a reactive gas, interacts with surfactants, antioxidants, and other compounds in the epithelial lining fluid (ELF). The compounds thought responsible for adverse pulmonary effects of inhaled NO_2 are the reaction products themselves or the metabolites of these products in the ELF. At the time of the 1993 AQCD for Oxides of Nitrogen, it was thought that inhaled NO_2 probably reacted with the water molecules in the ELF to form nitrous acid (HNO_2) and nitric acid (HNO_3). However, some limited data suggested that the absorption of NO_2 was linked to reactive substrates in the ELF and subsequent nitrite production. Since then, the reactive absorption of NO_2 has been examined in a number of studies (see Section 4.2.2). These studies have characterized the absorption kinetics and reactive substrates for NO_2 delivered to various sites in the respiratory tract. Researchers have attempted to obtain a greater understanding of how these complex interactions affect NO_2 absorption and NO_2-induced injury.

With respect to quantifying absolute NO_2 absorption, the following were reported in the 1993 AQCD for Oxides of Nitrogen. The principles of O_3 uptake were generally assumed applicable for NO_2 modeling studies. The results indicated that NO_2 is absorbed throughout the lower respiratory tract, but the major delivery site is the centriacinar region, i.e., the junction between the conducting and respiratory airways in humans and animals. Experimental studies have found that the total respiratory tract uptake in humans ranges from 72 to 92% depending on the study and the breathing conditions. The percent total uptake increases with increasing exercise level. In laboratory animals, upper respiratory tract uptakes ranged from as low as 25% to as high as 94% depending on the study, species, air flow rate, and mode of breathing (nasal or

oral). Upper respiratory tract uptake of NO_2 was found to decrease with increasing ventilation. Uptake during nasal breathing was determined to be significantly greater than during oral breathing.

AX4.2.1 Mechanisms of NO_2 Absorption

The ELF is the initial barrier against NO_2 delivery to the underlying epithelial cells. Postlethwait and Bidani (1990) suggested that acute NO_2 uptake in the lower respiratory tract was rate limited by chemical reactions of NO_2 with ELF constituents rather than by gas solubility in the ELF. Subsequently, Postlethwait et al. (1991) reported that inhaled NO_2 (10 ppm) does not penetrate the ELF to reach underlying sites and suggested that cytotoxicity may be due to NO_2 reactants formed in the ELF. Since then, the reactive absorption of NO_2 has been examined in a number studies that have sought to identify reactive substrates for NO_2 and quantify the absorption kinetics of NO_2 in the respiratory tract.

Postlethwait and Bidani (1994) concluded that the reaction between NO_2 and water does not significantly contribute to the absorption of inhaled NO_2. Uptake is a first-order process for NO_2 concentrations less than 10 ppm, is aqueous substrate-dependent, and is saturable. The absorption of inhaled NO_2 is thought to be coupled with free radical-mediated hydrogen abstraction to form HNO_2 and an organic radical (Postlethwait and Bidani, 1989, 1994). At physiologic pH, the HNO_2 subsequently dissociates to H^+ and nitrite (NO_2^-). The concentration of the resulting nitrite is thought insufficient to be toxic, so effects are thought to be due to the organic radical and/or the proton load. Nitrite may enter the underlying epithelial cells and blood. In the presence of red blood cells, nitrite is oxidized to nitrate (NO_3^-) (Postlethwait and Mustafa, 1981). Beyond cell susceptibility and the concentration of NO_2 in the lumen, site-specific injury was proposed to depend on rate of 'toxic' reaction product formation and the quenching of these products within the ELF. Related to the balance between reaction product formation and removal, it was further suggested that cellular responses may be nonlinear with greater responses being possible at low levels of NO_2 uptake versus higher levels of uptake. Since the ELF may vary throughout the respiratory tract, the uptake of inhaled NO_2 and reaction with constituents of the pulmonary ELF may be related to the heterogeneous distribution of epithelial injury observed from NO_2 exposure.

Postlethwait et al. (1995) sought to determine the absorption substrates for NO_2 in the ELF lavaged from male Sprague-Dawley rats. Since the bronchoalveolar lavage fluid (BALF) collected from the rats may be diluted up to 100-fold relative to the native ELF, the effect of concentrating the BAL fluid on NO_2 absorption was investigated. A linear association was found between the first-order rate constant for NO_2 absorption and the concentration of the BALF. This suggests that concentration of the reactive substrates in the ELF determines the rate of NO_2 absorption. The absorption due to specific ELF constituents was also examined in chemically pure solutions. Albumin, cysteine, reduced glutathione (GSH), ascorbic acid, and uric acid were hydrophilic moieties found to be active substrates for NO_2 absorption. Unsaturated fatty acids (such as oleic, linoleic, and linolenic) were also identified as active absorption substrates and thought to account for up to 20% of NO_2 absorption. Vitamins A and E exhibited the greatest reactivity of the substrates that were examined. However, the low concentrations of uric acid and vitamins A and E were thought to preclude them from being appreciable substrates in vivo. The authors concluded that ascorbate and GSH were the primary NO_2 absorption substrates in rat ELF. Postlethwait et al. (1995) also found that the pulmonary surfactant, dipalmitoyl phosphatidylcholine, was not an effective substrate for NO_2 absorption. Later, Connor et al. (2001) suggested that dipalmitoyl phosphatidylcholine may actually inhibit NO_2 absorption.

In a subsequent study, Velsor and Postlethwait (1997) investigated the mechanisms of acute epithelial injury from NO_2 exposure. The impetus for this work was to evaluate the supposition that NO_2 reaction products rather than NO_2 itself cause epithelial injury. Red blood cell membranes were immobilized to the bottom of Petri dishes, covered with a variety of well characterized aqueous layers, and exposed to gaseous NO_2 (10 ppm for 20 min). The study focused on the potential roles of GSH and ascorbic acid reaction products in mediating cellular injury. Based on negligible membrane oxidation when covered with only an aqueous phosphate buffer, the diffusive/reactive resistance of a thin aqueous layer clearly prevented direct interaction between NO_2 and the underlying membrane. The presence of unsaturated fatty acids was not observed to affect NO_2 absorption, but a sufficiently thin liquid layer was required for membrane oxidation to occur. Interestingly, membrane oxidation was not a simple monotonic function of GSH and ascorbic acid levels. The maximal levels of membrane oxidation were observed at low antioxidant levels versus null or high antioxidant levels. Glutathione and

ascorbic acid related membrane oxidation were superoxide and hydrogen peroxide dependent, respectively. The authors suggested that at the higher antioxidant concentrations, there was increased absorption of NO_2, but little secondary oxidation of the membrane because the reactive species (e.g., superoxide and hydrogen peroxide) generated during absorption were quenched. At the low antioxidant concentrations, there was a lower rate of NO_2 absorption, but oxidants were not quenched and so were available to interact with the cell membrane.

Kelly et al. (1996a) examined the effect of a 4-h NO_2 (2 ppm) exposure on antioxidant levels in bronchial lavage fluid (BLF) and BALF of 44 healthy nonsmoking adults (19-45 year, median 24 years). Subjects were randomly assigned to three groups and lavaged at either 1.5 h ($n = 15$), 6 h ($n = 15$), or 24 h ($n = 14$) after the NO_2 exposure. The baseline concentrations of uric acid and ascorbic acid were strongly correlated between the BLF and BALF within individuals ($r = 0.88$, $p < 0.001$; $r = 0.78$, $p = 0.001$; respectively), whereas the concentrations of GSH in the BLF and BALF were not correlated. Uric acid levels in both lavage fractions were significantly reduced at 1.5 h ($p < 0.04$), significantly increased at 6 h ($p < 0.05$), and back to baseline at 24 h postexposure. A statistically significant loss of ascorbic acid was also found in both lavage fractions at 1.5 h ($p < 0.05$). At 6 and 24 h postexposure, the ascorbic acid levels had returned to baseline. In contrast, GSH levels were significantly increased at both 1.5 h ($p < 0.01$) and 6 h ($p < 0.03$) in BLF. At 24 h postexposure, the GSH levels in BLF returned to baseline. Although GSH in BLF increased at 1.5 and 6 h postexposure, oxidized GSH levels remained similar to baseline in both BLF and BALF. No changes in BALF levels of GSH were observed at any time point.

The depletion of uric acid and ascorbic acid, but not GSH has also been observed with *ex vivo* exposure of human BALF to NO_2. Kelly et al. (1996b) collected BALF from male lung cancer patients ($n = 16$) and exposed the BALF ex vivo at 37°C to NO_2 (0.05 to 2.0 ppm; 4 h) or O_3 (0.05 to 1.0 ppm; 4 h). Kelly and Tetley (1997) also collected BALF from lung cancer patients ($n = 12$, 54 ± 16 years) and exposed the BALF *ex vivo* to NO_2 (0.05 to 1.0 ppm; 4 h). Both studies found that NO_2 depletes uric acid and ascorbic acid, but not GSH from BALF. Kelly et al. (1996b) noted a differential consumption of the antioxidants with uric acid loss being greater than that of ascorbic acid which was lost at a much greater rate than GSH. Kelly and Tetley (1997) found that the rates of uric acid and ascorbic acid consumption were correlated with their initial concentrations in the BAL fluid, such that higher initial antioxidant

concentrations were associated with a greater rate of antioxidant depletion. Illustrating the complex interaction of antioxidants, these studies also suggest that GSH oxidized by NO_2 may be again reduced by uric acid and/or ascorbic acid.

AX4.2.3 Regional and Total Respiratory Absorption of NO_2

There has been very limited work related to the quantification of NO_2 uptake since the 1993 AQCD for Oxides of Nitrogen. As a result, there is an abbreviated discussion here of some papers that were reviewed in the 1993 AQCD for Oxides of Nitrogen.

AX4.2.3.1 Dosimetry Models

There is a paucity of theoretical studies investigating NO_2 dosimetry. Like O_3, NO_2 is highly reactive in ELF and is not very soluble. An O_3 model has been utilized to predict the uptake of NO_2 in the lower respiratory tract of humans, rats, guinea pigs, and rabbits (Miller et al., 1982; Overton, 1984). In this model, there was a strong distinction between uptake and dose. Uptake referred to the amount of NO_2 being removed from gas phase per lung surface area ($\mu g/cm^2$), whereas, dose referred to the amount of NO_2 per lung surface area ($\mu g/cm^2$) that diffused through the ELF and reached the underlying tissues.

Miller et al. (1982) and subsequently Overton (1984) did not attempt to predict the amount of reactants in the ELF or the transport of reactants to the tissues. Rather, they focused mainly on the sensitivity of NO_2 tissue dose on NO_2 reaction rates in the ELF and the Henry's law constant. Reaction rates of NO_2 in the ELF were varied from zero, 50%, and 100% of the reaction rate for O_3 in ELF. The Henry's law constant was varied from half to double the Henry's law constant for NO_2 in water at 37 °C. Effects of species, lung morphology, and tidal volume (V_T) were also examined. In general, the model predicted that NO_2 is taken up throughout the lower respiratory tract. In humans, NO_2 uptake was fairly constant from the trachea to the first generation of respiratory bronchioles, beyond which uptake decreased with distal progression. The NO_2 tissue dose was highly dependent on the Henry's law constant and reaction rate in the ELF. In the conducting airways, the NO_2 tissue dose decreased as the Henry's law constant increased (i.e., decreased gas solubility), whereas the NO_2 tissue dose in the alveolar region increased. The site of maximal NO_2 tissue dose was fairly similar between species, ranging from the first generation of respiratory bronchioles in humans to the alveolar ducts in rats. In guinea pigs and rabbits, the maximal NO_2 tissue dose was predicted to occur in

the last generation of respiratory bronchioles. The simulations showed that exercise increases the NO_2 tissue dose in the pulmonary region relative to rest. Miller et al. (1982) also reported that increasing the NO_2 reaction rate decreased NO_2 tissue dose in the conducting airways, but had no effect on the dose delivered to the pulmonary region.

Simultaneously occurring diffusion and chemical reactions in the ELF have been suggested as the limiting factors in O_3 (Santiago et al., 2001) and NO_2 uptake (Postlethwait and Bidani, 1990). Hence, Miller et al. (1982) should have found an increase in the uptake of NO_2 in the conducting airways with increasing the rate of chemical reactions in the ELF. This increase in NO_2 uptake in the conducting airways would then lead to a reduction in the amount of NO_2 reaching and taken up in the pulmonary region. The Miller et al. (1982) model considered reactions of NO_2 with constituents in the ELF as protective in that these reactions reduced the flux of NO_2 to the tissues. Others have postulated that NO_2-reactants formed in the ELF, rather than NO_2 itself, could actually cause adverse responses (Overton, 1984; Postlethwait and Bidani, 1994; Velsor and Postlethwait, 1997).

More recently, Overton and Graham (1995) examined NO_2 uptake in an asymmetric anatomic model of the rat lung. The multiple path model of Overton and Graham (1995) allowed for variable path lengths from the trachea to the terminal bronchioles, whereas Miller et al. (1982) used a single or typical path model of the conducting airways. The terms dose and uptake were used synonymously to describe the amount of NO_2 gas lost from the gas phase in a particular lung region or generation by Overton and Graham (1995). Reactions of NO_2 in the ELF were not explicitly considered. Their simulations were conducted for rats breathing at 2 mL V_T at a frequency of 150 breaths per minute. The mass transfer coefficients of 0.173, 0.026, and 0.137 cm/sec were assumed for the upper respiratory tract, the tracheobronchial airways, and the pulmonary region, respectively. Uptake was predicted to decrease with distal progression into the lung. In general, the modeled NO_2 dose varied among anatomically equivalent ventilatory units as a function of path length from the trachea with shorter paths showing greater dose. A sudden increase in NO_2 uptake was predicted in the proximal alveolar region (PAR) which was due to the increase in the assumed mass transfer coefficient relative to the adjacent terminal bronchiole. Overton et al. (1996) showed that increasing the mass transfer coefficient of the tracheobronchial airways would decrease the dose to the PAR and vice versa. Additionally, the PAR dose would also be reduced by the more realistic modeling of

tracheobronchial airways expansion during inspiration versus the static condition employed by Overton and Graham (1995).

In summary, these modeling studies predict that the net NO_2 dose (NO_2 flux to air-liquid interface) gradually decreases distally from the trachea to the terminal bronchioles and then rapidly decreases in the pulmonary region. However, the tissue dose of NO_2 (NO_2 flux to liquid-tissue interface) is low in the trachea, increases to a maximum in the terminal bronchioles and the first generation of the pulmonary region, and then decreases rapidly with distal progression. The production of toxic NO_2-reactants in the ELF and the movement of the reactants to the tissues as not been modeled.

Experimental Studies of NO_2 Uptake

Upper Respiratory Tract Absorption

The nasal uptake of NO_2 has been experimentally measured in dogs, rabbits, and rats under conditions of unidirectional flow. Yokoyama (1968) reported 42.1 ± 14.9% (Mean ± StDev) uptake of NO_2 (4 to 41 ppm) in the isolated nasal passages of two dogs (3.5 L/min) and three rabbits (0.75 L/min) exposed to 7520 to 77,100 $\mu g/m^3$ (4 and 41 ppm) NO_2. Uptake did not appear to depend on the exposure concentration and was relatively constant over a 10 to 15 min period. Cavanagh and Morris (1987) measured uptakes of 28% and 25% uptake of NO_2 (76,000 $\mu g/m^3$; 40.4 ppm) in the noses of four naive and four previously exposed rats (0.10 L/min), respectively. Uptake was not affected by a 4-h prior exposure (naive versus previously exposed rats) to 76,000 $\mu g/m^3$ (40.4) ppm NO_2 and was constant over the 24-min period during which uptake was determined.

Kleinman and Mautz (1991) measured the penetration of NO_2 through the upper airways during inhalation in six tracheotomized dogs exposed to 1880 or 9400 $\mu g/m^3$ (1.0 or 5.0 ppm) NO_2. Uptake in the nasal passages was significantly greater at 1880 $\mu g/m^3$ (1.0 ppm) than at 9400 $\mu g/m^3$ (5.0 ppm), although the magnitude of this difference was not reported. The mean uptake of NO_2 (1880 $\mu g/m^3$; 1.0 ppm) in the nasal passages decreased from 55% to 40% as the ventilation rate increased from about 2 to 8 L/min. During oral breathing, uptake was not dependent on concentration. The mean oral uptake of NO_2 (1880 and 9400 $\mu g/m^3$; 1.0 and 5.0 ppm) decreased from 65% to 30% as the ventilation rate increased from 2 to 8 L/min.

Lower Respiratory Tract Absorption

Postlethwait and Mustafa (1989) investigated the effect of exposure concentration and breathing frequency on the uptake of NO_2 in isolated perfused rat lungs. To evaluate the effect of exposure concentration, the lungs were exposed to NO_2 (7520 to 37,600 μg/m^3; 4 to 20 ppm) while ventilated at 50 breaths/min with a V_T of 2.0 mL. To examine the effect of breathing frequency, the lungs were exposed to NO_2 (94,000 μg/m^3; 5 ppm) while ventilated at 30-90 breaths/min with a V_T of 1.5 mL. All exposures were for 90 min. The uptake of NO_2 ranged from 59 to 72% with an average of 65% and was not affected by exposure concentration or breathing frequency. A combined regression showed a linear relationship between NO_2 uptake and total inspired dose (25 to 330 μg NO_2). Illustrating variability in NO_2 uptake measurements, Postlethwait and Mustafa (1989) observed 59% NO_2 uptake in lungs ventilated at 30 breaths/min with a V_T of 1.5 mL, whereas, Postlethwait and Mustafa (1981) measured 35% NO_2 uptake for the same breathing condition. In another study, 73% uptake of NO_2 was reported for rat lungs ventilated 50 breaths/min with a V_T of 2.3 mL (Postlethwait et al., 1992). It should be noted that typical breathing frequencies are around 80, 100, and 160 breaths/min for rats during sleep, rest, and light exercise, respectively (Winter-Sorkina and Cassee, 2002). Hence, the breathing frequencies at which NO_2 uptake has been measured are lower than for rats breathing normally.

In addition to measuring upper respiratory tract uptakes, Kleinman and Mautz (1991) also measured NO_2 uptake in the dog lung. In general, there was about 90% NO_2 uptake in the lungs which was independent of ventilation rates from 3 to 16 L/min.

Total Respiratory Tract Absorption

Bauer et al. (1986) measured the uptake of NO_2 (560 μg/m^3; 0.3 ppm) in 15 adult asthmatics exposed for 30 min (20 min at rest, then 10 min exercising on a bicycle ergometer) via a mouthpiece during rest and exercise. There was a statistically significant increase in uptake from 72% during rest to 87% during exercise. The minute ventilation also increased from 8.1 L/min during rest to 30.4 L/min during exercise. Hence, exercise increased the dose rate of NO_2 by 5-fold in these subjects. In an earlier study of seven healthy adults in which subjects were exposed to a nitric oxide (NO)/NO_2 mixture containing 550 to 13500 μg/m^3 (0.29 to 7.2 ppm) NO_2 for brief (but unspecified) periods, Wagner (1970) reported that NO_2 uptake increased from 80% during normal respiration (V_T, 0.4 L) to 90% during maximal respiration (V_T, 2 to 4 L).

Kleinman and Mautz (1991) also measured the total respiratory tract uptake of NO_2 (9400 μg/m^3; 5 ppm) in female beagle dogs while standing at rest or exercising on a treadmill. The dogs breathed through a small face mask. Total respiratory tract uptake of NO_2 was 78% during rest and increased to 94% during exercise. In large part, this increase in uptake may be due to the increase in V_T from 0.18 L during rest to 0.27 L during exercise. Coupled with an increase in minute ventilation from 3.8 L/min during rest to 10.5 L/min during exercise, the dose rate of NO_2 was 3-fold greater for the dogs during exercise than rest.

Distribution and Elimination of NO_2 Products

As stated earlier, NO_2 absorption is coupled with nitrous acid (HNO_2) formation, which subsequently dissociates to H^+ and nitrite (NO_2^-). Nitrite enters the underlying epithelial cells and subsequently the blood. In the presence of red blood cells and possibly involving oxyhemoglobin, nitrite is oxidized to nitrate (NO_3^-) (Postlethwait and Mustafa, 1981). Nitrate may subsequently be excreted in the urine. There has been concern that inhaled NO_2 may lead to N-nitrosamine production, many of which are carcinogenic, since NO_2 can produce nitrite and nitrate (in blood). Nitrate can be converted to nitrite by bacterial reduction in saliva, the gastrointestinal tract, and the urinary bladder. Nitrite has been found to react with secondary amines to form N-nitrosamines. This remains speculative since nitrosamines are not detected in tissues of animals exposed by inhalation to NO_2 unless precursors to nitrosamines and/or inhibitors of nitrosamine metabolism are co-administered. Rubenchik et al. (1995) could not detect N-nitrosodimethylamine (NDMA) in tissues of mice exposed to 7.5 to 8.5 mg/m^3 NO_2 for 1 h. NDMA was found in tissues, however, if mice were simultaneously given oral doses of amidopyrine and 4-methylpyrazole, an inhibitor of NDMA metabolism. Nevertheless, the main source of NO_2 in the body is formed endogenously, and food is also a contributing source of nitrite from the conversion of nitrates. Thus, the relative importance of inhaled NO_2^- to N-nitrosamine formation has yet to be demonstrated.

Metabolism of inhaled NO_2 may also transform other chemicals that may be present in the body, in some cases into mutagens and carcinogens. Van Stee et al. (1995) exposed mice to approximately 37,600 μg/m^3 (20 ppm) $^{15}NO_2$ and to 1 g/kg morpholine simultaneously. N-nitrosomorpholine (NMOR), a nitrosamine that is a potent animal carcinogen, was found in the body of the exposed mice. Ninety-eight point four percent was labeled with ^{15}N that was

derived from the inhaled $^{15}NO_2$ and 1.6% was derived presumably from endogenous sources. Miyanishi et al. (1996) co-exposed rats, mice, guinea pigs and hamsters to NO_2 and various polycyclic aromatic hydrocarbons (PAHs) such as pyrene, fluorene, or anthracene. Nitro derivatives of these PAHs were excreted in the urine of co-exposed animals, which were found to be highly mutagenic in the Ames/S. typhimurium assay. Specifically, the nitrated metabolite of pyrene (1-nitro-6/8-hydroxypyrene and 1-nitro-3hydroxypyrene) was detected in the urine. Further studies indicated that these metabolites are nitrated by an ionic reaction in vivo after the hydroxylation of pyrene in the liver.

Extra-Pulmonary Effects of NO_2 and NO

Exposure to NO_2 produces a wide array of health effects beyond the confines of the lung. Thus, NO_2 and/or some of its reactive products penetrate the lung or nasal epithelial and endothelial layers to enter the blood and produce alteration in blood and various other organs. Effects on the systemic immune system were discussed above and the summary of other systemic effects is quite brief because the database suggests that effects on the respiratory tract and immune response are of greatest concern. A more detailed discussion of extrapulmonary responses can be found in U.S. Environmental Protection Agency (1993).

Body Weight, Hepatic, and Renal Effects

Conflicting results have been reported on whether NO_2 affects body weight gain in experimental animals as a general indicator of toxicity (U.S. Environmental Protection Agency, 1993). Newer subchronic studies show no significant effects on body weight in rats, guinea pigs, and rabbits exposed up to 7526 $\mu g/m^3$ (4 ppm) NO_2 (Tepper et al., 1993; Douglas et al., 1994; Fujimaki and Nohara, 1994).

Effects on the liver, such as changes in serum chemistry and xenobiotic metabolism, have been reported by various investigators to result from exposure to NO_2 (U.S. Environmental Protection Agency 1993). Drozdz et al. (1976) found decreased total liver protein and sialic acid, but increased protein-bound hexoses in guinea pigs exposed to 2000 $\mu g/m^3$ (1.05 ppm) NO_2, 8 h/day for 180 days. Liver alanine and aspartate aminotransferase activity was increased in the mitochondrial fraction but decreased in the cytoplasmic fraction of the liver. Electron micrographs of the liver showed intracellular edema and inflammatory and parenchymal degenerative changes.

No new studies on liver effects were located in the literature since the 1993 AQCD for Oxides of Nitrogen. Several older studies have shown changes in kidney function and xenobiotic metabolism in animals following NO_2, although no histopathological changes were reported.

Increases in urinary protein and specific gravity of the urine were reported by Sherwin and Layfield (1974) in guinea pigs exposed continuously to 940 $\mu g/m^3$ (0.5 ppm) NO_2 for 14 days. Proteinuria (albumin and alpha-, beta-, and gamma-globulins) was found in another group of animals when the exposure was reduced to 752 $\mu g/m^3$ (0.4 ppm) NO_2 for 4 h/day. However, differences in water consumption or in the histology of the kidney were not found. No new studies were located in the literature since the 1993 AQCD for Oxides of Nitrogen.

Brain Effects

There are several studies suggesting that NO_2 affects the brain. Decreased activity of protein metabolizing enzymes, increased glycolytic enzymes, changes in neurotransmitter levels (5-HT and noradrenaline), and increased lipid peroxidation, accompanied by lipid profile and antioxidant changes, have been reported (Farahani and Hasan, 1990, 1991, 1992; Sherwin et al., 1986; Drozdz et al., 1975). The U.S. Environmental Protection Agency (1993) concluded that "none of these effects have been replicated and all reports lack sufficient methodological rigor; thus, the implications of these findings, albeit important, are not clear and require further investigation".

A developmental neurotoxicity study by Tabacova et al.(1985) suggest that in utero exposure to NO_2 may result in postnatal neurobehavioral development changes as described in the section on reproductive and developmental toxicology.

Van Stee et al. (1983) reported NMOR production in mice gavaged with 1 g of morpholine/kg body weight per day and then exposed (5-6 h daily for 5 days) to 31,000 to 38,500 $\mu g/m^3$ (16.5 to 20.5 ppm) NO_2. The single site containing the greatest amount of NMOR was the gastrointestinal tract. In a later experiment, 98.4% of the NMOR found in the body of mice exposed to ~20 ppm (i.e., ~37 600 mg/m^3) $^{15}NO_2$ and to 1 g/kg morpholine was labeled with ^{15}N that was derived from the $^{15}NO_2$ to which the animals had been exposed by inhalation, and 1.6% was derived from $^{14}NO_2$ from presumably endogenous sources (Van Stee et. al., 1995).

Inhaled NO_2 may also be involved in the production of mutagenic (and carcinogenic) nitro derivatives of other co-exposed compounds, such as polycyclic aromatic hydrocarbons

(PAHs), via nitration reactions. Miyanishi et al. (1996) co-exposed rats, mice, guinea pigs and hamsters to 37,600 μg/m^3 (20 ppm) NO_2 and various PAHs (pyrene, fluoranthene, fluorene, anthracene, or chrysene). Nitro derivatives of these PAHs were excreted in the urine of these animals, which were found to be highly mutagenic in the Ames/*S. typhimurium* assay. Specifically, the nitrated metabolite of pyrene (1-nitro-6/8-hydroxypyrene and 1-nitro-3hydroxypyrene) was detected in the urine. Further studies indicated that these metabolites are nitrated by an ionic reaction in vivo after the hydroxylation of pyrene in the liver.

NO

The genotoxicity of NO has been studied both in vitro and in vivo (Arroyo et al., 1992; Nguyen et al., 1992) (see Table AX4.8). Overall, the synthesis of these older studies suggests that NO has some genotoxic potential; however, the effect is slight and to a lesser extent when compared to NO_2.

Effects of Mixtures Containing NO_2

Humans are generally exposed to NO_2 in a mixture with other air pollutants. A limitation of animal toxicity studies is the extrapolation of dose-response data from controlled exposures to NO_2 alone to air pollutant mixtures that are typically found in the environment. It is difficult to predict the effects of NO_2 in a mixture based on the effects of NO_2 alone. In order to understand how NO_2 is affected by mixtures of other air pollutants, studies are typically conducted with mixtures containing NO_2 and one or two other air pollutants, such as O_3 and/or H_2SO_4. The result of exposure to two or more pollutants may be simply the sum of the responses to individual pollutants (additivity), may be greater than the sum of the individual responses, suggesting some type of interaction or augmentation of the response (synergism) or may be less than additive (antagonism).

Animal toxicity studies have shown an array of interactions, including no interaction, additivity or synergism. Because no clear understanding of NO_2 interactions has yet emerged from this database, only a brief overview is provided here. A more substantive review can be found in U.S. Environmental Protection Agency (1993). There are animal studies, which have studied the effects of ambient air mixtures containing NO_2 or gasoline or diesel combustion exhausts containing NO_x. Generally these studies provide useful information on the mixtures, but lack NO_2-only groups, making it impossible to discern the influence of NO_2. Therefore, this

class of research is not described here, but is reviewed elsewhere (U.S. Environmental Protection Agency, 1993).

Simple Mixtures Containing NO_2

Most of the interaction studies have involved NO_2 and O_3. After subchronic exposure, lung morphology studies did not show any interaction of NO_2 with O_3 (Freeman et al., 1974) or with SO_2 (Azouley et al., 1980). Some biochemical responses to NO_2 plus O_3 display no positive interaction or synergism. For example, Mustafa et al. (1984) found synergism for some endpoints (e.g., increased activities of O_2 consumption and antioxidant enzymes), but no interaction for others (e.g., DNA or protein content) in rats exposed for 7 days. Ichinose and Sagai (1989) observed a species dependence in regard to the interaction of O_3 (752 $\mu g/m^3$, 0.4 ppm) and NO_2 (752 $\mu g/m^3$, 0.4 ppm) after 2 weeks of exposure. Guinea pigs, but not rats, had a synergistic increase in lung lipid peroxides. Rats, but not guinea pigs, had synergistic increases in antioxidant factors (e.g., non-protein thiols, vitamin C, glucose-6-phosphate dehydrogenase, GSH peroxidase). Duration of exposure can have an impact. Schlesinger et al. (1990) observed a synergistic increase in prostaglandin E_2 and $F_{2\alpha}$ in the lung lavage of rabbits exposed acutely for 2 h to 5640 $\mu g/m^3$ (3.0 ppm) NO_2 plus 588 $\mu g/m^3$ (0.3 ppm) O_3; the response appeared to have been driven by O_3. However, with 7 or 14 days of repeated 2-h exposures, only prostaglandin E_2 was decreased and appeared to have been driven by NO_2; there was no synergism (Schlesinger et al., 1991).

Using the infectivity model (see Section AX4.3.2.5 for protocol details), Ehrlich et al. (1977) found additivity after acute exposure to mixtures of NO_2 and O_3 and synergism after subchronic exposures. Exposure scenarios involving NO_2 and O_3 have also been performed using a continuous baseline exposure to one concentration or mixture, with superimposed short-term peaks to a higher level (Ehrlich et al., 1979; Gardner, 1980, 1982; Graham et al., 1987). Differences in the pattern and concentrations of the exposure are responsible for the increased susceptibility to pulmonary infection, without indicating clearly the mechanism controlling the interaction.

Some aerosols may potentiate response to NO_2 by producing local changes in the lungs that enhance the toxic action of co-inhaled NO_2. The impacts of NO_2 and H_2SO_4 on lung host defenses have been examined by Schlesinger and Gearhart (1987) and Schlesinger (1987a). In the former study, rabbits were exposed for 2 h/day for 14 days to either 564 $\mu g/m^3$ (0.3 ppm) or

1880 μg/m^3 (1.0 ppm) NO_2, or 500 μg/m^3 H_2SO_4 alone, or to mixtures of the low and high NO_2 concentrations with H_2SO_4. Exposure to either concentration of NO_2 accelerated alveolar clearance, whereas H_2SO_4 alone retarded clearance. Exposure to either concentration of NO_2 with H_2SO_4 resulted in retardation of clearance in a similar manner to that seen with H_2SO_4 alone. Using a similar exposure design but different endpoints, exposure of rabbits to 1800 μg/m^3 (1.0 ppm) NO_2 increased the numbers of PMNs in lavage fluid at all time points (not seen with either pollutant alone), and increased phagocytic capacity of AMs after two or six exposures (Schlesinger et al., 1987). Exposure to 564 μg/m^3 (0.3 ppm) NO_2 with acid, however, resulted in depressed phagocytic capacity and mobility. The NO_2/H_2SO_4 mixture was generally either additive or synergistic, depending on the specific cellular endpoint being examined.

Exposure to high levels of NO_2 ($\leq$9400 μg/m^3, 5.0 ppm) with very high concentrations of H_2SO_4 (1 mg/m^3) caused a synergistic increase in collagen synthesis rate and protein content of the lavage fluid of rats (Last and Warren, 1987; Last, 1989).

Complex Mixtures Containing NO_2

Although many studies have examined the response to NO_2 with only one additional pollutant, the atmosphere in most environments is a complex mixture of more than two materials. A number of studies have attempted to examine the effects of multi-component atmospheres containing NO_2, but as mentioned before, in many cases the exact role of NO_2 in the observed responses is not always clear. One study by Stara et al. (1980) deserves mention because pulmonary function changes appeared to progress after exposure ceased.

In the study by Stara et al. (1980), dogs were exposed for 68 months (16 h/day) to raw or photochemically reactive vehicle exhaust which included mixtures of NO_x: one with a high NO_2 level and a low NO level (1200 μg/m^3, 0.64 ppm, NO_2; 310 μg/m^3, 0.25 ppm, NO), and one with a low NO_2 level and a high NO level (270 μg/m^3, 0.14 ppm, NO_2; 2050 μg/m^3, 1.67 ppm, NO). At the end of exposure, the animals were maintained for about 3 years in normal indoor air. Numerous pulmonary functions, hematological and histological endpoints were examined at various times during and after exposure. The lack of an NO_2-only or NO-only group precludes determination of the nature of the interaction. Nevertheless, the main findings are of interest. Pulmonary function changes appeared to progress after exposure ceased. Dogs in the high NO_2 group had morphological changes considered to be analogous to human centrilobular

emphysema. Because these morphological measurements were made after a 2.5- to 3-year holding period in clean air, it cannot be determined with certainty whether these disease processes abated or progressed during this time. This study suggests progression of damage after exposure ends.

TABLE AX4.1. EFFECTS OF NITROGEN DIOXIDE (NO_2) ON OXIDANT AND ANTIOXIDANT HOMEOSTASIS[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
75 752 7,520	0.04 0.4 4.0	Continuous, 9 and 18 mos	M	8 wks	Rat (Wistar)	NPSHs increased at ≥0.4 ppm after 9 or 18 mos; GSH peroxidase activity increased after a 9-mo exposure to 4.0 ppm; G-6-P dehydrogenase was increased after a 9- and 18-mo exposure to 4.0 ppm; no effects on 6-P-G dehydrogenase , SOD disulfide reductase; some GSH S-transferase had decreased activities after 18-mo exposure to ≥0.4 ppm.	Sagai et al. (1984) Ichinose et al. (1983)
752	0.4	2 wks	NS	NS	Rat Guinea Pig	No effect on TBA reactants, antioxidants, or antioxidant enzyme activities.	Ichinose and Sagai (1989)
752 2,260 7,520	0.4 1.2 4.0	Continuous, 4 mos	M	13 wks	Rat (Wistar)	Duration dependent pattern for increase in activities of antioxidant enzymes; increase, peaking at wk 4 and then decreasing. Concentration-dependent effects.	Ichinose and Sagai (1982)
752-940	0.4-0.5	Continuous, 1.5 yrs	F	NS	Mouse (NS)	Growth reduced; Vitamin E (30 or 300 mg/kg diet) improved growth.	Csallany (1975)
940 1,880	0.5 1.0	Continuous, 17 mos	F	4 wks	Mouse (C57B1/6J)	At 1 ppm, GSH-peroxidase activity decreased in vitamin E-deficient mice and increased in Vitamin E- supplemented mice.	Ayaz and Csallany (1978)
1,880	1.0	4 h/day, 6 days	NS	NS	Rat (Sprague-Dawley)	Vitamin E-supplement reduced lipid peroxidation.	Thomas et al. (1967)
1,880 4,330 11,600	1.0 2.3 6.2	Continuous, 4 days	M	8 wks	Rat (Sprague-Dawley)	Activities of GSH reductase and G-6-P dehydrogenase increased at 6.2 ppm proportional to duration of exposure; plasma lysozyme and GSH peroxidase not affected at 6.2 ppm; no effects at 1.0 or 2.3 ppm.	Chow et al. (1974)
2,260 3,380	1.2 1.8	Continuous, 3 days	M	12 wks	Rat (Sprague-Dawley)	Increases in G-6-P dehydrogenase, isocitrate dehydrogenase, disulfide reductase, and NADPH cytochrome c reductase activities at 1.8 ppm only.	Lee et al. (1989, 1990)
3,760 18,800	2.0 10.0	3 days	M/F	5->60 days	Rat (Wistar) Guinea pig (Dunkin Hartley)	Decreased SOD activity in 21-day-old animals.	Azoulay-Dupuis et al. (1983)

TABLE AX4.1 (cont'd). EFFECTS OF NITROGEN DIOXIDE (NO_2) ON OXIDANT AND ANTIOXIDANT HOMEOSTASIS[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
3,760 7,500 18,800	2.0 4.0 10.0	14 days 10 days 7 days	M	12-24 wks	Rat (Wistar)	G-6-P dehydrogenase increased at ≥2 ppm; at 2 ppm, 14 days of exposure needed	Mochitate et al. (1985)
5,600	3.0	7 days	M/F	1 day to >8 wks	Rat (Sprague-Dawley)	Increased lipid peroxidation (TBA-reactive substances) with vitamin E deficiency.	Sevanian et al. (1982)
17,900	9.5	7 h/day, 5 days/wks, 6 mos	M	In utero and 6 mos	Rat (Fischer 344)	Increase in GSH reductase activity in younger rats and SDH peroxidase activity in older rats.	Mauderly et al. (1987)
5,600	3.0	4 days	M	NS	Rat (Sprague-Dawley)	No effects on parameters tested.	Mustafa et al. (1979)
13,200	7.0	4 days				Increase in lung weight, G-6-P dehydrogenase, GSH reductase, and GSH peroxidase activities. Increased lung weight, G-6-P dehydrogenase; and GSH reductase activities.	
18,800 28,200	10 15	4 days 1-7 days				Increase in lung weight, DNA content, G-6-P dehydrogenase, 6-P-G dehydrogenase, GSH reductase, disulfide reductase, GSH peroxidase, disulfide reductase, succinate oxidase, and cytochrome oxidase activities; no effect on lung protein	
7,520	4.0	3 h	M/F	21-33 yrs	Human	Decreased elastase inhibitory capacity and increased lipid peroxidation products in BAL of subjects not administered supplement of vitamin C and E prior to NO_2 exposure.	Mohsenin (1991)
9,400 18,800	5.0 10.0	Continuous, 24 h 7 days	M	NS	Rats (CD Cobs)	Changes in the GSH levels in blood and lung occurred in rats exposed for 24 h, but returned to normal after 7 days.	Pagani et al. (1994)

TABLE AX4.1 (cont'd). EFFECTS OF NITROGEN DIOXIDE (NO_2) ON OXIDANT AND ANTIOXIDANT HOMEOSTASIS[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
11,000	6.0	4 h/day, 30 days	F	NS	Mouse (NS)	Increase in GSH reductase and G-6-P dehydrogenase activities.	Csallany (1975)
28,000 53,000	15 28	7 days				Increase in GSH levels, G-6-P dehydrogenase, and GSH peroxidase activities.	
17,900	9.5	7 h/day, 5 days/wk, 24 mos	M	18 wks	Rat (Fischer 344)	Increase in GSH reductase activity in BAL.	Mauderly et al., (1990)
18,800	10.0	Continuous 3 days, 20 days	NS	NS	Rat (Fisher 344)	Decreased GSH/GSSG ratio in blood and BALF, but not in lung type II cells. Lipid peroxidation was decreased in type II cells at 3 days, but was similar to controls at 20 days. mRNA expression of the enzymes involved in the biosynthesis (γGCS and GS) was decreased at both time points. γGT (redox of GSH) mRNA expression was increased.	Hochscheid et al. (2005)
26,320	14.0	NS	NS	NS	Human	Rapid depletion of vitamin C, glutathione and vitamin E	Halliwell et al. (1992)

[a]Modified from US Environmental Protection Agency (1993).

M= Male

NPSHs= Nonprotein sulfhydryls.

GSH= Glutathione.

G-6-P dehydrogenase= Glucose-6-phosphate dehydrogenase.

6-P-G dehydrogenase= 6-phosphosgluconate dehydrogenase.

SOD= Superoxide dismutase.

F-Female.

NS= Not Stated.

NADP= Nicotinamide-adenine dinucleotide phosphate (reduced form).

TBA= Thiobarbituric acid

TABLE AX4.2. EFFECT OF NITROGEN DIOXIDE (NO_2) ON LUNG AMINO ACIDS, PROTEINS, AND ENZYMES[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species**	**Effects**	**References**
752 1,880 5,640 9,400	0.4 1.0 3.0 5.0	72 h	M	NS	Guinea Pig (Hartley)	No effect at 0.4 ppm; increase in BAL protein in vitamin C-depleted, but not normal, animals at 1.0 ppm and above.	Selgrade et al. (1981)
9,400	5.0	3 h				Increased BAL protein in vitamin C-depleted guinea pigs 15 h postexposure.	
752	0.4	Continuous, 1 wk				No effect on BAL protein.	
752	0.4	Continuous, 1 wk	M	NS	Guinea Pig	Increased protein content of BAL from vitamin-C-deficient guinea-pigs.	Sherwin and Carlson (1973)
752 2,260 7,520	0.4 1.2 4.0	1-14 wks	M	22-24 wks	Rat (Wistar)	Complex concentration and duration dependence of effects. Example: at 0.4 ppm, cytochrome P-450 levels decreased at 2 wks, returned to control level by 5 wks. At 1.2 ppm, cytochrome P-450 levels decreased initially, increased at 5 wks, and decreased at 10 wks. Effects on succinate-cytochrome c reductase also.	Takahashi et al. (1986)
940 1,880	0.5 1.0	6 h/day, 5 days/wk, 4 wks	M	NS	Rat (Fischer 344)	0.5 ppm; increase in urinary hydroxylysine output starting during wk 1; BAL hydroxylysine level, angiotensin-converting enzyme level, and BAL protein content unchanged. 1.0 ppm: gradual increase in urinary hydoxylysine output, becoming significant the week after exposure ended; BAL hydroxylysine level lower following exposure and 4 wks postexposure; andiotensin-converting enzyme level increased.	Evans et al. (1989)
1,880 14,100 28,200 47,000 56,400	1.0 7.5 15 25 30	6 h/day, 2 days				Concentration dependent increase in urinary hydroxylysine output and BAL hydroxyxlysine content, but only significant at ≥7.5 ppm and 15 ppm, respectively; angiotensin-converting enzyme levels and BAL protein increased in highest-exposed groups.	

TABLE AX4.2 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON LUNG AMINO ACIDS, PROTEINS, AND ENZYMES[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species**	**Effects**	**References**
1,880 9,400	1.05.0	7h/day, 5 days/wk, up to 15 wks	M/F	14-16 wks	Rat (Fischer 344)	Change in BAL and tissue levels of enzymes early in exposure, resolved by 15 wks.	Gregory et al. (1983)
752 2,260 7,520	1.2 1.2 4.0	7 days	M	10 wks	Rat (Wistar)	Decrease in levels of cytochrome P-450 at 1.2 ppm.	Mochitate et al. (1984)
3,760	2.0	1, 2, or 3 wks	M	NS	Guinea pig	Increased lactate dehydrogenase (LDH) content of the lower lobes of the lung	Sherwin and Carlson (1973)
1,504 9,400 18,800	0.8 5 10	1 or 3 days	? [check]	? [check]	Rat ([check])	BAL protein content significantly increased in a concentration- and exposure duration-dependent manner, with the change becoming significant at 5 ppm for 3 days and at 10 ppm for ≥1 day of exposure.	Muller et al. (1994)
3,760 7,520 18,800	2.0 4.0 10	14 days 10 days 7 days	M	12-24 wks	Rats (Wistar)	Increase activity of lung glycolytic enzymes.	Mochitate et al. (1985)
5,640	3.0	7 days	M/F	8 wks	Rat (Sprague-Dawley)	Various changes in lung homogenate protein and DNA content and enzyme activities, changes more severe in vitamin E-deficient rats.	Elsayed and Mustafa (1982)
6,770 13,500 20,300 27,100	3.6 7.2 10.8 14.4	24 h 12 h 8 h 6 h	M	10-12 wks	Rat (Sprague-Dawley)	Increased BAL protein ≥7.2 ppm.	Gelzleichter et al. (1992)
7,520 18,800	4.0 10	10 days 7 days	M	21-24 wks	Rat (Wistar)	Initial decrease in lung protein content followed by an increase; changes on microsomal enzyme activities.	Mochitate et al. (1984)
7,520 18,800 47,000	4.0 10 25	6 h/day 5 days/wk, 7, 14, and 21 days	M	NS	Rat (Wistar)	Increased gamma-glutamyl transferase on days 14 and 21; no consistent effect on alkaline phosphatase, LDH, or total protein.	Hooftman et al. (1988)

TABLE AX4.2 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON LUNG AMINO ACIDS, PROTEINS, AND ENZYMES[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species**	**Effects**	**References**
8,100	4.5	16 hrs	M/F	NS	Guinea pig (Hartley)	Increased lung wet weight, alterations in lung antioxidant levels in Vitamin C- deficient animals.	Hatch et al. (1986)
9,020	4.8	3 hrs	M			Increased lung lavage fluid protein content in vitamin C-deficient animals.	
9,020	4.8	8 h/day, 7 days	M	8 wks	Mouse (Swiss Webster)	No significant changes in lung homogenate parameters.	Mustafa et al. (1984)
9,400	5.0	14-72 h	F	NS	Mouse (NS)	Increase in lung protein (14 to 58 h) by radioactive label incorporation.	Csallany (1975)
9,400	5.0	2 wks	M	5 wks	Rat (Fischer 344)	Increased amounts of the tryptophan metabolites and xanthurenic and kynurenic acids excreted in urine during wk 2 of exposure, but had returned to normal levels by wk 4.	Suzuki et al. (1988)
9,400	5.0	6 h/day, 6 days	NS	NS	Mice	Modest increase in albumin in BAL; no effect on LDH or lysosomal enzyme peroxidase.	Rose et al. (1989)
9,400-47,000	5.0-25.0	Continuous, 7 days	M	10-11 wks	Rat (Sprague-Dawley)	Concentration-related increase in collagen synthesis rate; 125% increase in rats exposed to 5.0 ppm.	Last et al. (1983)
9,400 37,600 94,000	5.0 20.0 50.0	3 h	NS	NS	Rabbit (New Zealand)	Benzo [a] pyrene hydroxylase activity of tracheal mucosa not affected.	Palmer et al. (1972)
9,400	5.0	Continuous, 1, 3, or 7 days	M	NS	Rat (Sprague-Dawley)	Increased BAL protein at 3 days (day 7 not measured); increased (120% collagen synthesis at 7 days (not measured other days).	Last & Warren (1987)
15,000	8.0	Continuous, 14 days	F	NS	Mouse (NS)	Increase in lung protein.	Csallany (1975)
17,900	9.5	7 h/day, 5 days/wk, 6 mos	M	In utero and 6 mos	Rat (Fischer 344)	Increase in BAL alkaline phosphatase, acid phosphatase, and LDH in older rats only.	Mauderly et al. (1987)
17,900	9.5	7 h/day, 5 days/wk, 24 mos	M	18 wks	Rat (Fischer 344)	Increase in BAL levels of LDH and alkaline phosphatase activities and in collagenous peptides.	Mauderly et al. (1990)

TABLE AX4.2 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON LUNG AMINO ACIDS, PROTEINS, AND ENZYMES[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species**	**Effects**	**References**
18,800	10	24 h or 7 days	M	NS	Rat (CD cobs)	Protein content of BALF increased significantly in rats after only 24 h. BALF elastase activity was not affected. concentration-dependent increase in α-1 proteinase inhibitor content after 24 h of exposure, but not with longer exposures.	Pagani et al. (1994)
18,800	10	Continuous, 14 days	M	8 wks	Rat (Wistar)	Changes in several enzymes in whole lung homogenates.	Sagai et al. (1982)
18,800 37,600 56,400 75,200	10 20 30 40	4 h	M	NS	Rat (Long Evans)	Increased activities of various enzymes, sialic acid, and BAL protein; attenuation by high dietary levels of vitamin E.	Guth and Mavis (1985, 1986)

[a]Modified from US Environmental Protection Agency (1993).

NS = Not Stated

LTB_4 = Leukotriene B4

LDH = Lactate Dehydrogenase

M=Male

F= Female

BAL= Bronchoalveolar lavage

TABLE AX4.3. EFFECTS OF NITROGEN OXIDE (NO_2) ON THE IMMUNE SYSTEM OF ANIMALS[a]

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
940 188 base + 470, 940, or 1,880 peak	0.5 0.1 base + 0.25, 0.5, or 1.0 peak	Continuous Continuous base + 3 h/day, 5 days/wk peak for 1, 3, 6, 9, 12 mos	NS	NS	Mouse	Suppression of splenic T and B cell responsiveness to mitogens variable and not related to concentration or duration, except for the 940 µg/m³ continuous group, which had a linear decrease in PHA-induced mitogenesis with NO_2 duration.	Maigetter et al. (1978)
470	0.25	7 h/day, 5 days/wk, 7wks	F	6 wks	Mouse (AKR/cum)	Reduced percentage of total T-cell population and trend towards reduced percentage of certain T-cell subpopulations; no reduction of mature T cells or natural killer cells.	Richters and Damji (1988)
470	0.25	7 h/day, 5 days/ wk, 36 wks	F	5 wks	Mouse (AKR/cum)	Reduced percentage of total T-cell population and percentages of T helper/inducer cells on days 37 and 181.	Richters and Damji (1990)
658	0.35	7 h/day, 5 days/ wk, 12 wks	M	6 wks	Mouse (C57BL/6J)	Trend towards suppression in total percentage of T-cells. No effects on percentages of other T-cell subpopulations.	Richters and Damji (1988)
752 3,010	0.4 1.6	24 h/day 4 wks	M	7 wks	Mouse (BALB/c)	Decrease in primary PFC response at ≥752 µg/m³. Increase in secondary PFC response at 3010 µg/m³.	Fujimaki et al. (1982)
940 base + 2,820 peak	0.5 base + 1.5 peak	22 h/day, 7 days/wk base + 6 h/day, 5 days/wk peak for 1, 3, 13, 52, 78 wks	M	10 wks	Rat (Fischer 344)	No effect on splenic or circulatory B or T cell response to mitogens. After 3 weeks of exposure only, decrease in splenic natural killer cell activity. No histological changes in lymphoid tissues.	Selgrade et al. (1991)
940 base + 3,760 peak 3,760	0.5 base + 2.0 peak	24 h/day, 5 days/wk base + 1 h/day, 5 days/wk peak for 3 mos	M	6 wks	Mouse (CD-1)	Vaccination with influenza A2/Taiwan virus after exposure. Decrease in serum neutralizing antibody; hemagglutination inhibition antibody titers unchanged. Before virus challenge, NO_2 exposure decreased serum IgA and increased IgG1, IgM, and IgG2; after virus, serum IgA unchanged and IgM increased.	Ehrlich et al. (1975)

TABLE AX4.4. EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
94 base + 3,760	0.05 base + 2.0 peaks	3 h base + three 15-min peaks	NS	NS	Human	No effects at 0.05 ppm NO_2 with peaks; trend ($p < 0.07$) towards AMs losing ability to inactivate influenza virus at 0.6 ppm.	Frampton et al. (1989)
1,130	0.6	3 h					
188 1,880 9,400 37,600	0.1 1.0 5.0 20	1 h	NS	NS	Rat (Sprague-Dawley) (in vitro)	At 5.0 ppm: increase in LTB_4; concentration-related decrease in SOD production in AMs at ≥1.0 ppm; increase in LDH in AMS at 5.0 and 20 ppm	Robison et al. (1990)
376 940 3760	0.2 0.5 2.0			Gestation 12 wks	Rat (Brown-Norway)	Reactive oxygen species generation from alveolar macrophages was significantly suppressed in NO_2 exposed weanling animals; no changes in reactive oxygen generating capability in the embryonic exposed animals.	Kumae and Arakawa (2006)
940	0.5	Continuous, 24 wks	NS	NS	Mouse	No effects on AM morphology at 0.5 ppm continuous or 0.1 ppm base + peak.	Aranyi et al. (1976)
188 base + 1,880 peak	0.1 base + 1.0 peak	Continuous base + 3-h peak, 5 days/wk, 24 wks				After 21 weeks of exposure to 2.0 ppm continuous or 0.5 ppm base + peak, morphological changes were identified, such a loss of surface processes, appearance of fenestrae, bleb formation, and denuded surface areas.	
3,760	2.0	Continuous, 33 wks					
940 base+ 3,760 peak	0.5 base + 2.0 peak	Continuous base + 1-h peak, 5 days/wk, 33 wks					

TABLE AX4.4 (cont'd). EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
560 1,880	0.3 1.0	2 h/day 2, 6, 13 days	M	NS	Rabbit (New Zealand)	Decreased phagocytic ability of AMs at 0.3 ppm after 2 days of exposure; increased at 1.0 ppm after 2 days of exposure; no effect on cell number or viability; random mobility reduced at 0.3 ppm only; no effects after 6 days of exposure.	Schlesinger (1987)
560 1,880	0.3 1.0	2 h/day up to 14 days	M	NS	Rabbit (New Zealand)	Increase in alveolar clearance.	Schlesinger and Gearhart (1987)
560 1,880 5,640 18,800 560 1,880	0.3 1.0 3.0 10 1.0 10	2 h 2 h/day, 14 days	M	NS	Rabbit (New Zealand)	Concentration-related acceleration in clearance of particles from lung with the greatest increase at two lowest concentrations, effects from repeated exposures similar to those seen after acute exposures to same concentrations.	Vollmuth et al. (1986)
940	0.5	0.5, 1, 5 and 10 days exposure	NS	NS	Rat (NS)	Superoxide production in alveolar macrophages from BALF, stimulated by phorbol myrisate acetate (PMA), was decreased after 0.5 days of exposure, and continued to be depressed after 1, 5, and 10 days.	Robinson et al. (1993)
940 base + 2,820 peak 3,760 base + 11,300	0.5 base + 1.5 peak 2.0 base +6.0 peak	Base 22 h/day, 7 days/wk + two 1-h peaks, 5 days/wk, 6 wks	M	1 day and 6 wks	Rat (Fischer 344)	Trend towards increase in number of AMs and cell volume in younger animals; increase in number of AMs and cell volume in older rats.	Crapo et al. (1984) Chang et al. (1986)

TABLE AX4.4 (cont'd). EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
1,000 2,500 5,000	0.5 1.3 2.7	Continuous, 28 days	M	6 wks	Rat (Wistar)	Increase in AMs in highest exposed group; no effects noted in 2 lowest exposure groups.	Rombout et al. (1986)
1,880 3,760 7,520	1.0 2.0 4.0	24 h/day, 12 wks			Guinea pig (NS) Rat (NS)	IgE-mediated histamine release from lung mast cells was enhanced in guinea pigs, but not rats exposed to 4.0 ppm. No effect observed at lower concentrations.	Fujimaki and Nohara, 1994
1,880 9,400 28,200	1.0 5.0 15	6 h/day, 2 days	NS	4-6 wks	Mouse (CD1)	Exposure-related decrease in AM phagocytosis from 1.0-5.0 ppm, decrease was not further affected by 15 ppm.	Rose et al. (1989)
1,880 3,760 7,520	1.0 2.0 4.0	24 h/day, 12 wks			Guinea pig (NS) Rat (NS)	IgE-mediated histamine release from lung mast cells was enhanced in guinea pigs, but not rats exposed to 4.0 ppm. No effect observed at lower concentrations.	Fujimaki and Nohara, (1994)
1,880 28,200 45,120	1.0 + 0.9 ppm No 15 24	7 h/day, 5 days/wks for 11 or 22 exposures	NS	NS	Rat (Long Evans)	Stimulated clearance of particles from lung at lowest concentration, but decreased clearance rate at two highest concentrations.	Ferin and Leach (1977)
1,880 9,400	1.0 5.0	7 h/day, 5 days/wks	M/F	14-16 wks	Rat (Fischer 344)	Accumulation of AMs. Superimposed peak exposures produced changes that may persist with continued exposures.	Gregory et al. (1983)
1,880 base + 9,400 peaks	base + 5.0 peaks	Base 7 h/day, 5 days/wks; two 1.5-h peaks/day; 15 wks					
2,440-32,000	1.3-17	NS ("acute")	F	NS	Rat (Sprague-Dawley)	Decreased production of superoxide anion radical.	Amoruso et al. (1981)

TABLE AX4.4 (cont'd). EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
3,760 19,000	2.0 10	3 days	M/F	5, 10, 21, 45, 55, 60, and >60 days	Guinea pig (Dunkin Hartley) Rat (Wistar)	Newborns were less affected than adults when AMs were tested for SOD levels.	Azoulay-Dupuis et al. (1983)
3,760	2.0	8 h/day, 5 days/wk, 6 mo	M/F	3-4 yrs	Baboon	Impaired AM responsiveness to migration inhibitory factor.	Green and Schneider (1978)
3,760	2.0	4 h	NS	NS	Human	Decreased phagocytosis and superoxide anion release.	Devlin et al. (1992)
5,000	2.7	24 h	M	6 wks	Rat (Wistar)	Increase in number of AMs.	Rombout et al. (1986)
5,640-30,100	3-6	3 h	NS	NS	Dog (Beagle)	Enhanced swelling of AMs.	Dowell et al. (1971)
6,770 22,700	3.6 12.1	1 h 2 h	F	NS	Rat (Sprague-Dawley) (in vitro)	Enhanced macrophage agglutination with concanavalin A at both concentrations tested.	Goldstein et al. (1977)
7,520 19,000 47,000	4 10 25	6 h/day, 7, 14, or 21 days	M	NS	Rat (Wistar)	Changes in morphology at all concentrations; increase in number of AMs at ≥10 ppm; phagocytic capacity reduced after 14 and 21 days of exposure to 25 ppm.	Hooftman et al. (1988)
7,520	4.0	10 days		19-23 wks		Increase in number of AMs; no increase in PMNs; increased metabolic activity, protein, and DNA synthesis; all responses peaked on day 4 and returned to normal on day 10.	Mochitate et al. (1986)

TABLE AX4.4 (cont'd). EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
7,520 15,000	4.0 8.0	Up to 10 days	NS	NS	Rat (Fischer 344)	Increase in number of AMs at both concentrations, reaching a peak on day 3 and 5; no increase in number of PMNs; decrease in AM viability throughout exposure period. Suppression of phagocytic activity after 7 days of exposure to 4 ppm and after 5 days of exposure to 8 ppm; returned to normal value at 10 days. Decrease in superoxide radical production, but at 4 ppm, the effect became significant on days 3, 5, and 10; at 8 ppm, the effect was significant at all time periods tested.	Suzuki et al. (1986)
9,400	5.0	7 days	F	NS	Mouse (CD-1)	No effect on phagocytic activity.	Lefkowitz et al. (1986)
9,400 28,200	5 15	3 h after infection with parainfluenza 3 virus	NS	NS	Rabbit (New Zealand)	AMs lost resistance to challenge with rabbit pox virus after exposure to 15 ppm.	Acton and Myrvik (1972)
9,400 18,800 28,200	5 10 15	3 h	M F[b]	NS	Humans (in vitro exposure)	No change in cell viability, release of neutrophil chemotactic factor, or interleukin-1.	Pinkston et al. (1988)
9,400-113,000	5-60	3 h	NS	NS	Rabbit (New Zealand)	Inhibition of phagocytic activity.	Gardner et al. (1969) Acton and Myrvik (1972)
13,200	7.0	24 h	NS	NS	Rabbit	Increased rosette formation in AMs treated with lipase.	Hadley et al. (1977)
17,900	9.5	7 h/day; 5 days/wk; 18-22 mo	M	18 wks	Rat (Fischer 344)	No effect on long-term clearance of radiolabeled tracer particles.	Mauderly et al. (1990)

TABLE AX4.4 (cont'd). EFFECTS OF NITROGEN DIOXIDE ON ALVEOLAR MACROPHAGES

NO_2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**Reference**
18,800	10	Continuous 7 days	NS	NS	Rat (NS)	High influx of PMNs in the lung (BALF) after 24 h of exposure, reversed for macrophages; no change in the lymphocyte population.	Pagani et al. (1994)
19,000	10	35 days	NS	NS	Guinea pig	63% increase in epithelial cells positive for macrophage congregation.	Sherwin et al. (1968)
19,000	10	4 h	F	NS	Mouse (Swiss)	Increase in total pulmonary cells in animals infected with some species of bacteria.	Jakab (1988)
19,000 47,000	10 25	24 h	M	12-13 wks	Rat (Sprague-Dawley)	Decreased phagocytosis at 25 ppm only.	Katz and Laskin (1976)

[a]NS = Not stated.
AMs = Alveolar macrophages.
LTB_4 = Leukotriene B_4.
LDH = Lactate dehydrogenase.
M = Male
F = Female
SOD = Superoxide dismutase.
PMNs = Polymorphonuclear leukocytes.
[b]Only one female used in study.

TABLE AX4.5. EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
100 base + 188 peak	0.05 base + 0.1 peak	Continuous, base + twice/day 1 h peaks, 5 days/wk for 15 days	F	NS	Mouse (CD-1)	*Streptococcus sp.*	No effect.	Gardner (1980, 1982) Graham et al. (1987)
940 base + 1,880 peak	0.5 + peak						Increased mortality.	
2,256 base + 4,700 peak	1.2 base + 2.5 peak						Increased mortality.	
376 base + 1,504 peak	0.2 base + 0.8 peak	23 h/day, 7 days/wk base+ twice daily 1 h peaks, 5 days/wk for 1 yr	F	6-8 wks	Mouse (CD-1)	*Streptococcus sp.*	Peak plus baseline caused significantly greater mortality than baseline.	Miller et al. (1987)
564-940	0.3-0.5	Continuous, 3 mos	F	4 wks	Mouse (ICR:JCL)	A/PR/8 virus	High incidence of adenomatous proliferation peripheral and bronchial epithelial cells; NO_2 alone and virus alone caused less severe alterations.	Motomiya et al. (1973)
		Continuous, 6 mos					No enhancement of effect of NO_2 and virus.	

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
940	0.5	Intermittent, 6 or 18 h/ day, up to 12 mos Continuous, 90 days	F	NS	Mouse (Swiss)	*K. pneumoniae*	Increased mortality after 6 mos intermittent exposure or after 3, 6, 9, or 12 mos continuous exposure, increased mortality was significant only in continuously exposed mice.	Ehrlich and Henry (1968)
940-1,880	0.5-1.0	Continuous, 39 days	F	NS	Mouse (ICR, dd)	A/PR/8 virus	Increased susceptibility to infection.	Ito (1971)
18,800	10	2 h/day, 1, 3, and 5 days						
940-52,700	0.5-28	Varied	F	NS	Mouse (CD-1)	*Streptococcus sp.*	Increase mortality with increased time and concentration; concentrations is more important than time.	Gardner et al. (1977 a,b) Coffin et al. (1977)
940	0.5	3 h/day, 3 mos	F	6-8 wks	Mouse (CD_2 F_1, CD-1)	*Streptococcus sp.*	Increase in mortality with reduction in mean survival time.	Ehrlich et al. (1979)
940 1,880 2,820	0.5 1.0 1.5	24 h/day, 7 days/wk, 3 mos	F	NS	Mouse (CF-1)	*K. pneumoniae*	Significant increase in mortality after 3-day exposure to 5.0 ppm; no effect at other concentrations, but control mortality very high.	McGrath and Oyervides (1985)
9,400	5.0	3 days						

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
940 1,880 3,760 9,400	0.5 1.0 2.0 5.0	4 h	M/F	8-10 wks	Mouse (C57BL/6N)	*Mycoplasma pulmonis*	Decrease in intrapulmonary killing only at 5.0 ppm.	Davis et al. (1991, 1992)
1,880 4,324 12,408	1.0 2.3 6.6	17 h	M	NS	Mouse (Swiss)	*S. aureus* after exposure	No difference in number of bacteria deposited, but at the two highest concentrations, there was a decrease in pulmonary bactericidal activity of 6 and 35%, respectively; no effect at 1.0 ppm	Goldstein et al. (1974)
1,880 4,700 9,400 18,800	1.0 2.5 5.0 10.0	4 h	F	NS	Mouse (Swiss)	*S. aureus*	Injection with corticosteroids increased NO_2-induced impairment of bactericidal activity at ≥2.5 ppm.	Jakab (1988)
1,880	1.0	48 h	M	NS	Mouse (Swiss Webster)	*Streptococcus sp. S. aureus*	Increased proliferation of *Streptococcus* in lung of exposed mice but no effect with *Streptococcus*	Sherwood et al. (1981)
1,800 5,640	1.0 3.0	3 h	F	5-6 wks	Mouse (CD-1)	*Streptococcus sp.*	Exercise on continuously moving wheels during exposure increased mortality at 3.0 ppm.	Illing et al. (1980)

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
1,880 4,700 9,400	1.0 2.5 5.0	6 h/day, 6 days	NS	4-6 wks	Mouse (CD-1)	Cytomegalovirus	Increase in virus susceptibility at 5.0 ppm only.	Rose et al. (1988, 1989)
2,820-94,000	1.5-50	2 h	NS	NS	Mouse (NS) Hamster (NS) Monkey (Squirrel)	*K. pneumoniae*	Increased mortality in mice, hamsters, and monkeys at ≥3.5, ≥35, and 50 ppm NO_2, respectively	Ehrlich (1980)
2,820	1.5	Continuous or intermittent, 7 h/day, 7 days/wk, up to 15 days	F	NS	Mouse (CD-1)	*Streptococcus sp.*	After 1 wk, mortality with continuous exposure was greater than that for intermittent after 2 wks, no significant difference between continuous and intermittent exposure.	Gardner et al. (1979) Coffin et al. (1977)
6,580	3.5						Increased mortality with increased duration of exposure; no significant difference between continuous and intermittent exposure; with data adjusted for total difference in C × T, mortality essentially the same.	

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
2,820 base + 8,100 peak	1.5 base + 4.5 peak	Continuous 64 h, then peak for 1, 3.5, or 7 h, then continuous 18 h base	F	NS	Mouse (CD-1)	*Streptococcus sp.*	Mortality increased with 3.5- and 7 h single peak when bacterial challenge was after an 18 h baseline exposure.	Gardner (1980) Gardner (1982) Graham et al. (1987)
8,100	4.5	1, 3.5, or 7 h					Mortality proportional to duration when bacterial challenge was immediate, but not 18 h postexposure.	
2,820	1.5	7 h/day, 4, 5, and 7 days	NS	NS	Mouse	*Streptococcus sp.*	Elevated temperature (32°C) increased mortality after 7 days.	Gardner (1982)
3,570 7,140 13,200 17,200 27,800	1.9 3.8 7.0 9.2 14.8	4 h	M	NS	Mouse (NS)	*S. aureus*	Physical removal of bacteria unchanged by exposure. Bactericidal activity decreased by 7, 14, and 50%, respectively, in three highest NO_2-exposed groups.	Goldstein et al. (1973)
2,820-9,400	1.5-5.0	3 h	F	6-10 wks	Mouse (CF-1, $CD2F_1$)	*Streptococcus sp.*	Increased mortality in mice exposed to ≥2.0 ppm	Ehrlich et al. (1977) Ehrlich (1980)

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
2,820 4,700 6,580 9,400 18,800 28,200	1.5 2.5 3.5 5.0 10 15	2 h	NS	6-8 wks	Mouse (Swiss Webster)	*K. pneumoniae*	No effect at 1.5 or 2.5 ppm; increased mortality at 3.5 ppm and above. Increase in mortality when *K. pneumoniae* challenge 1 and 6 h after 5 or 10 pm NO_2 exposure; when *K. pneumoniae* challenge 27 h following NO_2 exposure, effect only at 15 ppm.	Purvis and Ehrlich (1963) Ehrlich (1979)
3,760	2.0	1.5 h/day, 5 days/wk for 1, 2, and 3 wks	NS	2 wks	Hamster (Golden Syrian) (in vitro)	A/PR/8/34 influenza virus	Peak virus production in tracheal explants occurred earlier.	Schiff (1977)
4,700 7,500 9,400 18,800 28,200	2.5 4.0 5.0 10 15	4 h	F	NS	Mouse (Swiss)	*S. aureus*, *Proteus mirabilis, Pasteurella pneumotropica*	Concentration-related decrease in bactericidal activity at ≥4.0 ppm with S. aureus when NO_2 exposure after bacterial challenge; when NO_2 exposure was before challenge, effect at 10 ppm; NO_2 concentrations >5.0 ppm required to affect bactericidal activity for other tested microorganisms.	Jakab (1987, 1988)

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
9,400	5.0	Continuous, 2 mos	M	NS	Monkey (Squirrel)	*K. pneumoniae* or A/PR/8 influenza virus	Increased viral-induced mortality (1/3). Increase in *Klebsiella*-induced mortality (2/7); no control deaths.	Henry et al. (1970)
18,800	10	Continuous, 1 mo					Increased virus-induced mortality (6/6) within 2-3 days after infection; no control deaths. Increase in *Klebsiella*-induced mortality (1/4), no control deaths.	
9,400 18,880	5.0 10	4 h	M/F	6-10 wks	Mouse (C57B16N, C3H/HeN)	*Mycoplasma pulmonis*	NO_2 increased incidence and severity of pneumonia lesions and decreased the number of organisms needed to induce pneumonia; no effect on physical clearance; decreased mycoplasmal killing and increased growth; no effect on specific IgM in serum; C57B1/6N mice generally more sensitive than C3H/HeN mice. At 10 ppm, one strain (C57B1/6N) of mice had increased mortality.	Parker et al. (1989)

TABLE AX4.5 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON SUSCEPTIBILITY TO INFECTIOUS AGENTS[a]

NO_2 Concentration								
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Infective Agent**	**Effects**	**References**
18,800 28,200 65,800 94,000	10 15 35 50	2 h	M/F	NS	Monkey (Squirrel)	*K. pneumoniae*	Clearance of bacteria from lungs of 10-, 15-, and 35-ppm groups delayed or prevented. All three animals in highest exposed group died.	Henry et al. (1969)
9,400	5	?	?	?	Mice	Parainfluenza (murine sendei virus)	Altered the severity but not the course of the infection	Jakab (1988)

[a]Modified from US Environmental Protection Agency (1993)

F = Female.

M = Male.

NS = Not stated.

K. pneumoniae = Klebsiella pneumoniae

S. aureus = Staphylococcus aureus.

C × T = The product of concentration and time

TABLE AX4.6. EFFECT OF NITROGEN DIOXIDE (NO_2) ON HEMATOLOGICAL PARAMETERS[a]

NO^2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
94	0.05	Continuous 90 days	NS	NS	Rat	No effect on blood hemoglobin or RBCs.	Shalamberidze (1960)
677	0.36	1 wk	NS	NS	Guinea Pig	Increase of red blood cell D-2,3-diphosphoglycerate	Mersch et al. (1973)
940-1,500 +	0.5-0.8 +	Continuous 1 to 1.5 mos	M/F	4 wks	Mouse (ICR:JCL)	Addition of 50 ppm CO to NO_2 failed to affect carboxyhemoglobin.	Nakajima and Kusumoto (1970)
1500	0.8	Continuous, 5 days	M	7 wks	Mouse (ICR)	No effect on methemoglobin.	Nakajima and Kusumoto (1968)
1,880	1.0	Continuous, 16 mos	M	NS	Monkey (Squirrel)	No effect on hematocrit or hemoglobin with NO_2 and influenza exposure.	Fenters et al. (1973)
1,800 9,400	1.0 5.0	Continuous, 18 mos	M	NS	Dog (Mongrel)	No changes in hemoglobin or hematocrit..	Wagner et al. (1965)
1,880-56,400	1-30	18 h	NS	NS	Mouse (NS)	Concentration-related increase in methemoglobin and nitrosylhemoglobin	Case et al. (1979)
2,400-5,640	1.3-3.0	2 h/day, 15 and 17 wks	NS	NS	Rabbit (NS)	Decreased RBCs.	Mitina (1962)
3,760	2.0	Continuous, 14 mos	M/F M	NS	Monkey (Macaca speciosa) Rat (Sprague-Dawley)	With or without NaCl (330 µg/m³): polycythemia with reduced mean corpuscular volume and normal mean corpuscular hemoglobin.	Furiosi et al. (1973)
3,760	2.0	Continuous, up to 6 wks	M	8 wks	Rat (Wistar)	No effect on hemoglobin, hematocrit or RBC count; no methemoglobin was observed. Azoulay et al. (1978)	Azoulay et al. (1978)
7,520	4.0	1-10 days	NS	NS	Rat (NS)	Increase in RBC sialic acid.	Kunimoto et al. (1984)

TABLE AX4.6 (cont'd). EFFECT OF NITROGEN DIOXIDE (NO_2) ON HEMATOLOGICAL PARAMETERS[a]

NO^2 Concentration							
µg/m³	**ppm**	**Exposure**	**Gender**	**Age**	**Species (Strain)**	**Effects**	**References**
7,520	4.0	NS	NS	NS	NS	Decrease in RBCs.	Mochitate and Miura (1984)
9,400-75,200	5-40	1 h	F	4 mos	Mouse (JCL:ICR)	No increase in methemoglobin. Increased nitrite and especially nitrate.	Oda et al. (1981)
18,800	10	2 h/day, 5 days/wk, up to 30 wks	F	6-8 wks	Mouse (BALB/c)	Small decrease in hemoglobin and mean corpuscular hemoglobin concentration.	Holt et al. (1979)

[a]Modified from US Environmental Protection Agency (1993).

NS = Not stated.

RBCs = Red blood cells.

M = Male.

F = Female.

CO = Carbon monoxide.

NaCl = Sodium chloride.

TABLE AX4.7. EFFECTS OF NITRIC OXIDE WITH IRON AND ON ENZYMES AND NUCLEIC ACIDS

Effect	Reference
Sodium nitroprusside (NO donor) mobilizes iron from ferritin	Reif and Simmons (1990)
Modulation of arachidonic acid metabolism via interference with iron	Kanner et al. (1991, 1992)
Inhibition of aconitase (an enzyme in the Krebs cycle, and also complex 1 and 2 of the respiratory chain)	Hibbs et al. (1988) Persson et al. (1990) Stadler et al. (1991)
Permanent modification of hemoglobin, possibly via deamination	Moriguchi et al. (1992)
Deamination of DNA	Wink et al. (1991)
DNA strand breaks	Nguyen et al. (1992)
Inhibition of DNA polymerase and ribonucleotide reductase	Lepoivre et al. (1991) Kwon et al. (1991)
Antimitogenic; inhibition of T cell proliferation in rat spleen cells	Fu & Blankenhorn (1992)
Inhibition of DNA synthesis, cell proliferation, and mitogenesis in vascular tissue	Nakaki et al. (1990)
Inhibition of mitogenesis and cell proliferation (vascular smooth muscle cells)	Garg and Hassid (1989)
Adenosine diphosphate ribosylation is stimulated by NO-generating agents	Nakaki et al. (1990)

TABLE AX4.8A. GENOTOXICITY OF NO_2 IN VITRO AND IN PLANTS

Test Organism	End Point	Exposure	Comments	Results	Reference
Salmonella TA100	Mutations	6-10 ppm, 40 mins		+	Isomura et al. (1984)
Salmonella TA100	Mutations	10-15 ppm, 6 h	Concentrations >10 ppm were bacteriotoxic	+	Victorin and Ståhlberg (1988)
Salmonella TA100 and TA102	Mutations	Bubbling of 10-90 ppm through bact. susp., 30 mins as above		–	Kosaka et al. (1985)
Salmonella TA100	SOS repair	Bubbling of 10-90 ppm through bact. susp., 30 mins	Effect not considered solely attributed to nitrite in suspension. No effect seen with NO gas.	+	Kosaka et al. (1985)
E. coli, WP2	Mutations	Bubbling of 10-90 ppm through bact. susp., 30 mins		+	Kosaka et al. (1986, 1987)
E. coli	SOS repair	Bubbling of 10-90 ppm through bact. susp., 30 mins		+	Kosaka et al. (1986, 1987)
Bacillus subtilis spores	Mutations	500 ppm, 2-3 h		+	Sasaki et al. (1980)
V79 hamster cells	Chromatid-type aberrations, SCE	10-100 ppm, 10 mins	Effect shown not to be solely due to nitric acid or nitrite. No effect if cells not washed with Hank's salt solution prior to exposure	+	Tsuda et al. (1981)
V79 hamster cells	SCE	2-3 ppm, 10 mins		+	Shiraishi and Bandow (1985)
Don hamster cells	Mutations (8-G resistance)	2-3 ppm, 10 mins	Slight response	–	Isomura et al. (1984)
V79 hamster cells	DNA single-strand breaks	10 ppm, 20 mins	Effect not due to formation of nitrite	+	Görsdorf et al. (1990)
Tradescantia	Micronuclei in pollen	5 ppm, 24 h		+	Ma et al. (1982)
Tradescantia	Mutations in stamen hair	50 ppm, 6 h		+	Schairer et al. (1979)

Source: Victorin et al. (1994).

TABLE AX4.8B. GENOTICITY OF NO_2 IN VIVO

Test Organism	End Point	Exposure	Result	Reference
Drosophila	Recessive lethals	500-7000 ppm, 1 h	–	Inoue et al. (1981)
Drosophila	Somatic mutations (wing spot test)	50-280 ppm, 2 days	–	Victorin et al. (1990)
Rats	Mutations in lung cells (oubain res.)	50-560 ppm, >12 days	+	Isomura et al. (1984)
Rats	Chromosome aberrations in lung cells	27 ppm, 3 h	+	Isomura et al. (1984)
Mice	Chromosome aberrations in lymphocytes and spermatocytes	0.1-10 ppm, 6 h	–	Gooch et al. (1977)
Mice	Micronuclei in bone marrow	20 ppm, 23 h	–	Victorin et al. (1990)

Source: Victorin (1994).

TABLE AX4.8C. GENOTOXICITY OF NO IN VITRO AND IN VIVO

Test Organism	End Point	Exposure	Result	Reference
Salmonella TA100	Mutations	25-30 ppm, 40 min	+	Isomura et al. (1984)
Salmonella	SOS repair	Bubbling of 10-90 ppm	–	Kosaka et al. (1985)
Don hamster cells	Mutations (8-AG resistance)	2-3 ppm, 10 min	+	Isomura et al. (1984)
V79 hamster cells	DNA single-strand breaks	500 ppm, 30 min	–	Görsdorf et al. (1990)
TK 6 human cells	Mutations, DNA single-strand breaks	Injection of 0.12-0.38 ml NO gas/ml of culture medium, 1 h	+	Nguyen et al. (1992)
Salmonella TA1535	Mutations	30 min to 5-90 ppm	+	Arroyo et al. (1992)
Rats	Mutations in lung cells (oubain res.)	27 ppm, 3 h	–	Isomura et al. (1984)

Source: Victorin (1994); Arroyo et al. (1992) added.

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AX5. CHAPTER 5 ANNEX – CONTROLLED HUMAN EXPOSURE STUDIES OF NITROGEN OXIDES

AX5.1 INTRODUCTION

This annex summarizes the effects of nitrogen oxides (NO_x) on human volunteers exposed under controlled conditions. The goal is to review the scientific literature on human clinical studies of NO_x exposure published since the 1993 Air Quality Criteria Document (AQCD) for Oxides of Nitrogen (U.S. Environmental Protection Agency, 1993). Summary findings from the 1993 AQCD are provided below. The primary focus will be on nitrogen dioxide because it is the most abundant NO_x species in the atmosphere and there are few human studies of exposure to other NO_x species.

The following are the conclusions drawn from the review of clinical studies of nitrogen oxide exposure in the 1993 criteria document.

1. Nitrogen dioxide causes decrements in lung function, particularly increased airway resistance in healthy subjects at concentrations exceeding 2.0 ppm for 2 h.
2. Nitrogen dioxide exposure results in increased airway responsiveness in healthy, nonsmoking subjects exposed to concentrations exceeding 1.0 ppm for exposure durations of 1 hour or longer.
3. Nitrogen dioxide exposure at levels above 1.5 ppm may alter numbers and types of inflammatory cells in the distal airways or alveoli, but these responses depend upon exposure concentration, duration, and frequency. Nitrogen dioxide may alter function of cells within the lung and production of mediators that may be important in lung host defenses.
4. Nitrogen dioxide exposure of asthmatics causes, in some subjects, increased airway responsiveness to a variety of provocative mediators, including cholinergic and histaminergic chemicals, SO_2 and cold air. However, the presence of these responses appears to be influenced by the exposure protocol, particularly whether or not the exposure includes exercise.
5. Modest decrements in spirometric measures of lung function (3 to 8%) may occur in some asthmatics and COPD patients under certain NO_2 exposure conditions.
6. Nitric acid levels in the range of 50 to 200 ppb may cause some pulmonary function responses in adolescent asthmatics, but not in healthy adults. Other commonly occurring NO_x species do not appear to cause any pulmonary function responses at concentrations expected in the ambient environment, even at higher levels than in worst-case scenarios. However, not all nitrogen oxides acid species have been studied sufficiently.

7. No association between lung function responses and respiratory symptom responses were observed. Furthermore, there is little evidence of a concentration-response relationship for changes in lung function, airway responsiveness, or symptoms at the NO_2 levels that are reviewed here.

In the summary and integration chapter of the 1993 NO_x criteria document, one of the key health effects of most concern at near ambient concentrations of NO_2 was increases in airway responsiveness of asthmatic individuals after short-term exposures. The 1993 AQCD notes the absence of a concentration-response relationship for NO_2 exposure and airways responsiveness in asthmatics. For example, most responses to NO_2 that had been observed in asthmatics occurred at concentrations between 0.2 and 0.5 ppm. However, other studies showed an absence of effects on airways responsiveness at much higher concentrations, up to 4 ppm. Since 1993, additional studies suggest that exposure to low concentrations of NO_2, either alone or in combination with other pollutants such as SO_2, may enhance allergen responsiveness in asthmatic subjects.

In the years since the preparation of the 1993 AQCD, many studies from a variety of disciplines have convincingly demonstrated that exposure to particulate air pollution increases the risk for cardiovascular events. In addition, a number of epidemiological studies have shown associations between ambient NO_2 levels and adverse cardiovascular outcomes, at concentrations well below those shown to cause respiratory effects. However, to date there remain very few clinical studies of NO_2 that include endpoints relevant to cardiovascular disease.

AX5.1.1 Considerations in Controlled Human Exposure Studies

Strengths and Limitations of Controlled Human Studies

The database for air pollution risk assessment arises from four investigative approaches: epidemiology, animal toxicology, in vitro studies, and human inhalation studies. Each possesses advantages but also carries significant limitations. For example, the epidemiological investigation examines exposures in the "real world" but struggles with the realities of conducting research in the community, where cigarette smoking, socioeconomic status, occupational exposures, meteorological variability, and exposure characterization are important confounders. Outcomes are often evaluated from available health or mortality records or from administered questionnaires, all of which have inherent limitations. Sophisticated measures of

physiological responses are often not practical in studies involving large populations, although they may be used in panel studies. In contrast, inhalation studies in animals allow precision in quantifying exposure duration and concentration, measurement of a wide variety of physiologic, biochemical, and histological endpoints, and examination of extremes of the exposure-response relationship. Often, however, interpretation of these studies is constrained by difficulty in extrapolating findings from animals to humans, especially when exposure concentrations are unrealistically high.

Controlled, quantitative studies of exposed humans offer a third approach (Frampton et al., 2006). Human clinical studies attempt to engineer laboratory atmospheric conditions relevant to ambient pollutant atmospheres, with careful control of concentrations, duration, timing, and other conditions which may impact responses. These studies provide the opportunity to measure symptoms and physiological markers of health effects that result from breathing the atmospheres. The carefully controlled environment allows investigators to identify responses to individual pollutants, to characterize exposure-response relationships, to examine interactions among pollutants, and to study the effects of other variables such as exercise, humidity, or temperature. Susceptible populations can participate, including individuals with acute and chronic respiratory and cardiovascular diseases, with appropriate limitations based on subject comfort and protection from risk. Endpoint assessment traditionally has included symptoms and pulmonary function, but more recently a variety of markers of pulmonary, systemic, and cardiovascular function have been used to assess pollutant effects.

Human clinical studies have limitations. For practical and ethical reasons, studies must be limited to relatively small groups, to short durations of exposure, and to pollutant concentrations that are expected to produce only mild and transient responses. Findings from the short-term exposures in clinical studies may provide limited insight into the health effects of chronic or repeated exposures.

Specific issues of protocol design in human clinical studies have been reviewed (Frampton et al., 2006), and will not be considered further here, except in the context of specific studies of NO_2 exposure described in the following pages.

Assessing the Findings from Controlled Human Studies

In clinical studies, humans are the species of interest, so findings have particular relevance in risk assessment. However, the utility of clinical studies in risk assessment is

tempered by the obvious need to avoid adverse health effects of the study itself. This usually means selecting subjects that are not the most susceptible to the pollutant being studied. Furthermore, clinical studies depend on outcome markers with variable relevance or validation as markers of true health effects. The statement from the American Thoracic Society, "What constitutes an adverse health effect?" (American Thoracic Society, 2000) addresses issues relevant to selection and interpretation of outcome markers in clinical studies.

The 1993 NO_2 AQCD included a description of key outcome measures that had been in use to that date. These included primarily respiratory outcomes, including pulmonary function tests such as spirometry, lung volumes, and airways resistance, and tests of pulmonary clearance of inhaled aerosols. A brief description of bronchoalveolar lavage was also included, which had come into use prior to 1993 to assess airway inflammation and changes in the epithelial lining fluid in response to NO_2 exposure.

AX5.2 EFFECTS OF NITROGEN DIOXIDE IN HEALTHY SUBJECTS

Table AX5.1 summarizes the key clinical studies of NO_2 exposure in healthy subjects since 1993, with a few key studies included prior to that date. Figure AX5.1 summarizes the findings of these studies of airway inflammatory responses in relation to the total exposure to NO_2, expressed as ppm-minutes. Studies that did not include a proper control air exposure were not included, and studies using multiple daily exposures were not included. All of the studies portrayed in Figure AX5.1 involved intermittent exercise, and no attempt was made to adjust the exposure metric for varying intensity and duration of exercise.

AX5.3 THE EFFECTS OF NITROGEN OXIDE EXPOSURE IN SENSITIVE SUBJECTS

Table AX5.2 summarizes studies of potentially sensitive subjects. The potential for NO_2 exposure to enhance responsiveness to allergen challenge in asthmatics deserves special mention. Several recent studies, summarized in Table AX5.3, have reported that low-level exposures to NO_2, both at rest and with exercise, enhance the response to specific allergen challenge in mild asthmatics.

These recent studies involving allergen challenge suggest that NO_2 may enhance the sensitivity to allergen-induced decrements in lung function, and increase the allergen-induced

airway inflammatory response. Figure AX5.2 categorizes the allergen challenge studies as "positive", i.e. showing evidence for increased responses to allergen in association with NO_2 exposure, or "negative", with the exposure metric expressed as ppm-min. In comparing Figure AX5.2 with Figure AX5.1, it can be seen that enhancement of allergic responses in asthmatics occurs at exposure levels more than an order of magnitude lower than those associated with airway inflammation in healthy subjects. The dosimetry difference is even greater when considering that the allergen challenge studies were generally performed at rest, while the airway inflammation studies in healthy subjects were performed with intermittent exercise.

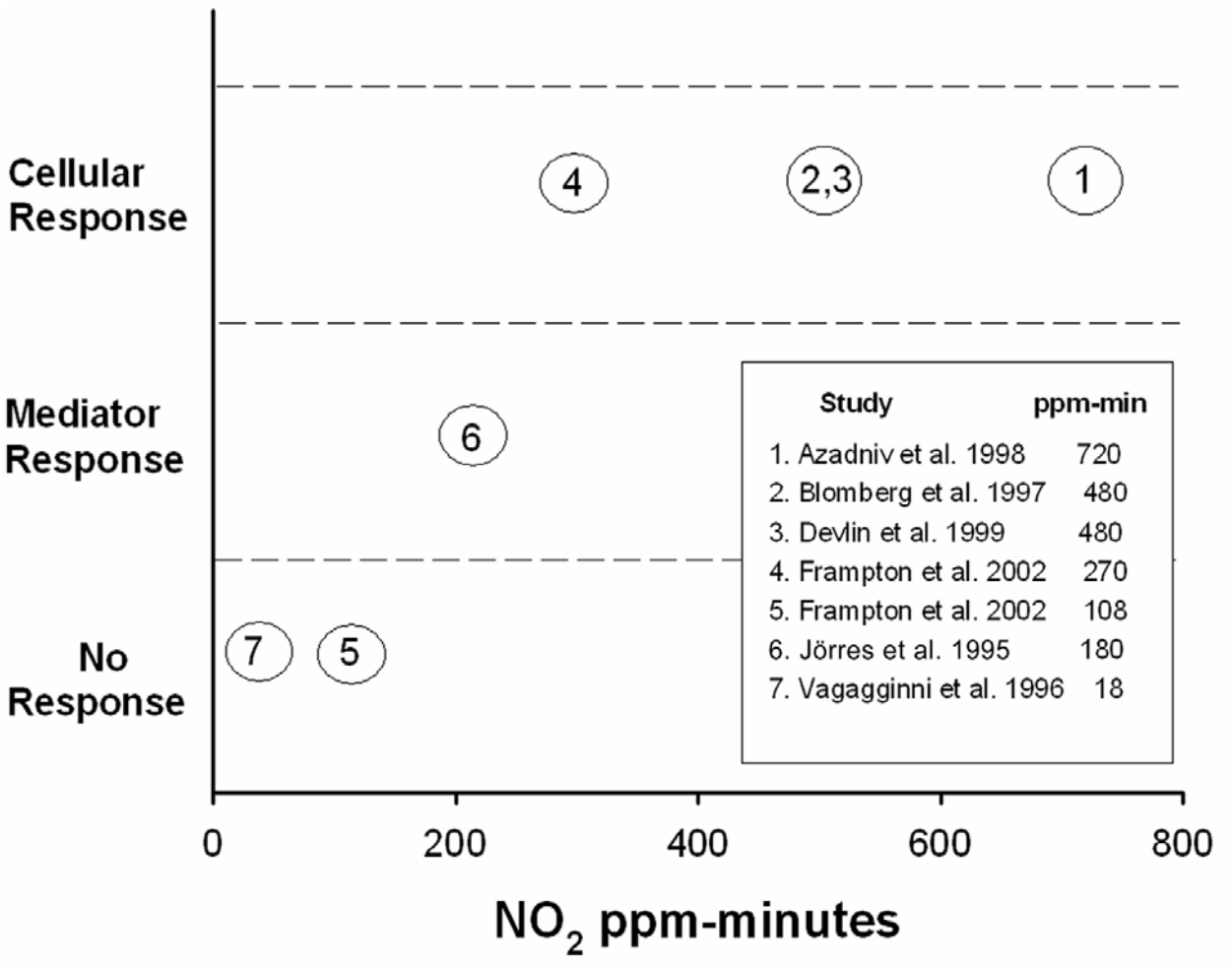

Figure AX5.1. Airway inflammation in response to NO_2 inhalation in healthy subjects.

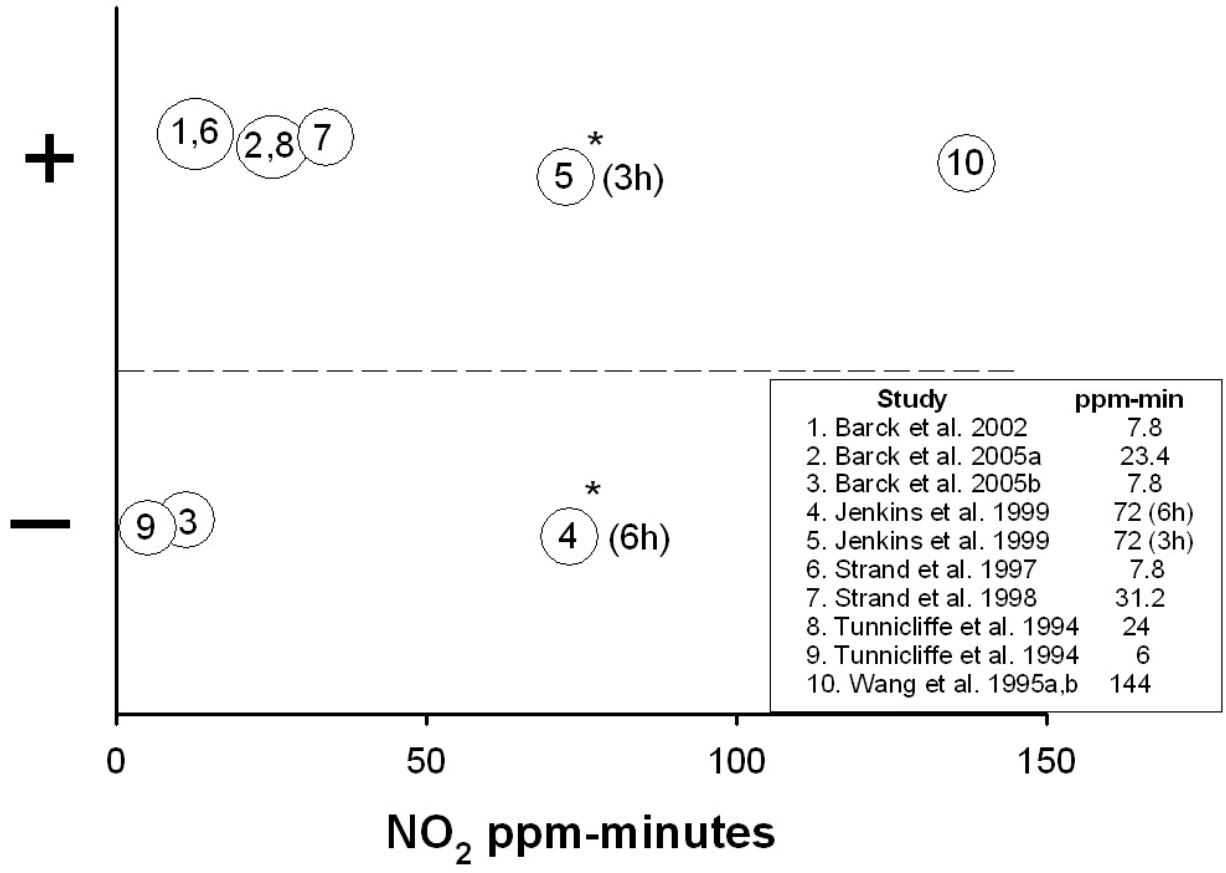

Figure AX5.2. Effects of NO_2 inhalation on allergen challenge in subjects with asthma. +: Significant effect of NO_2. -: No significant effect of NO_2. *: Exposures included intermittent exercise.

AX5.4 EFFECTS OF MIXTURES CONTAINING NITROGEN OXIDES

Table AX5.4 summarizes human clinical studies of NO_2-containing mixtures or sequential exposures that are most relevant to ambient exposure scenarios.

TABLE AX5.1. CLINICAL STUDIES OF NO_2 EXPOSURE IN HEALTHY SUBJECTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Avissar et al. (2001)	Rochester, NY, USA	21 healthy nonsmokers	Measurements of extracellular glutathione peroxidase (eGPx) activity and protein levels in epithelial lining fluid from NO_2 exposure study described in Frampton et al. (2002) (see below).	No effects of NO_2 exposure on eGPx activity and protein concentrations. (Ozone exposure decreased eGPx activity and protein concentrations.)	NO_2 up to 1.5 ppm for 3 hours did not deplete this mode of antioxidant defense in the epithelial lining fluid.
Azadniv et al. (1998)	Rochester, NY, USA	2 studies, 12 healthy nonsmokers in each	Air vs. 2 ppm NO_2 for 6 h with intermittent exercise. Phase 1: BAL 18 h after exposure; Phase 2: BAL immediately after exposure.	Increased BAL neutrophils, decreased blood CD8+ and null T lymphocytes 18 h after exposure. No effects on symptoms or lung function.	2 ppm NO_2 for 6 h caused mild inflammation.
Blomberg et al. (1997)	Sweden	30 healthy nonsmokers	Air vs. 2 ppm NO_2 for 4 h, with intermittent exercise	Increased neutrophils and interleukin-8 in bronchial wash. Increases in specific lymphocyte subsets in BAL fluid. Symptoms/lung function not reported.	2 ppm NO_2 for 4 h caused airway inflammation.
Blomberg et al. (1999)	Sweden	12 healthy nonsmokers	Air vs. 2 ppm NO_2 for 4 h on 4 days, with intermittent exercise.	After 4 days of NO_2, increased neutrophils in bronchial wash but decreased neutrophils in bronchial biopsy. 2% decrease in FEV_1 after first exposure to NO_2, attenuated with repeated exposure. Symptoms not reported.	Decreased lung function, not confirmed in other studies at t his concentration. Conflicting information on airway inflammation.
Devlin et al. (1999)	Chapel Hill, North Carolina, USA	8 healthy nonsmokers	Air and 2.0 ppm NO_2 for 4 h with intermittent exercise.	Increased bronchial lavage neutrophils, IL-6, IL-8, $alpha_1$-antitrypsin, and tissue plasminogen activator. Decreased alveolar macrophage phagocytosis and superoxide production. No effects on pulmonary function. Symptoms not reported.	2 ppm NO_2 for 4 h caused airway inflammation.
Drechsler-Parks (1995)	Santa Barbara, CA, USA	8 older healthy nonsmokers	4 2-h exposures with intermittent exercise: air, 0.60 ppm NO_2, 0.45 ppm O_3, and 0.60 ppm NO_2 + 0.45 ppm O_3.	Significant reduction in cardiac output during exercise, estimated using noninvasive impedance cardiography, with NO_2 + O_3. Symptoms and pulmonary function not reported.	Suggests cardiac effects of NO_2 + O_3. Small number of subjects limits statistical power, has not been replicated.

TABLE AX5.1 (cont'd). CLINICAL STUDIES OF NO_2 EXPOSURE IN HEALTHY SUBJECTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Frampton et al. (1991)	Rochester, NY, USA	39 healthy nonsmokers	3 protocols, all for 3 h with control air exposure: 1) continuous 0.06 ppm NO_2, 2) baseline 0.05 ppm NO_2 with peaks of 2.0 ppm, and 3) continuous 1.5 ppm NO_2.	No symptoms or direct effects on pulmonary function. Increased airways responsiveness to carbachol after 1.5 ppm NO_2.	Evidence for increased nonspecific airways responsiveness with NO_2 as low as 1.5 ppm for 3 h.
Frampton et al. (2002)	Rochester, NY, USA	21 healthy nonsmokers	Exposure to air, 0.6, 1.5 ppm NO_2 for 3 h with intermittent exercise.	Dose-related decrease in hematocrit, hemoglobin, blood lymphocytes, and T lymphocytes. Mild increase in neutrophils recovered in bronchial portion of BAL fluid. In vitro viral challenge of bronchial epithelial cells showed increased cytotoxicity after 1.5 ppm NO_2. No effects on symptoms or pulmonary function.	Indicates NO_2 causes airway inflammation below 1.5 ppm for 3 h. Suggest subtle effects on red blood cells, possibly RBC destruction (hemolysis).
Gong et al. (2005)	Downey, CA, USA	6 healthy nonsmokers and 18 ex-smokers with COPD	2 h exposures with intermittent exercise to: 1) air, 2) 0.4 ppm NO_2, 3) 200 $\mu g/m^3$ concentrated ambient particulate matter (CAPs), 4) NO_2 + CAPs	Reduced maximum mid-expiratory flow rate and oxygen saturation with CAPs exposures; no effects of NO_2 alone or additive effect with CAPs.	Exposures not fully randomized. Small number of healthy subjects limits interpretation for healthy group.
Helleday et al. (1994)	Sweden	8 healthy smokers, 8 healthy nonsmokers	3.5 ppm NO_2 for 20 min with 15 min exercise. BAL 24 h after exposure compared with non-exposure control BAL.	Different inflammatory cell increases in smokers and nonsmokers. No effects on symptoms. Pulmonary function not reported.	Lack of control air exposure with exercise is problematic.
Helleday et al. (1995)	Sweden	24 healthy nonsmokers, 8 in each of 3 groups	Bronchoscopic assessment of mucociliary activity: 1) 45 min after 1.5 ppm NO_2 for 20 min, 2) 45 min after 3.5 ppm NO_2 for 20 min, and 3) 24 h after 3.5 ppm NO_2 for 4 h.	Complete abolition of mucociliary activity 20 min after NO_2; increased activity 24 h after NO_2. Symptoms/pulmonary function not reported.	No true air control exposure, order of procedures not randomized, subjects not blinded.

TABLE AX5.1 (cont'd). CLINICAL STUDIES OF NO_2 EXPOSURE IN HEALTHY SUBJECTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Jörres et al. (1995)	Germany	8 healthy nonsmokers & 12 mild asthmatics	Air or 1 ppm NO_2 exposure for 3 h with intermittent exercise.	In asthmatics, 2.5% decrease FEV_1 after NO_2 vs. 1.3% decrease after air, $p = 0.01$. FEV_1 decreased 20% in 1 subject after NO_2. No significant lung function effect in healthy subjects. Changes in eicosanoids (more pronounced in asthmatics), but not inflammatory cells, in BAL fluid.	Lung function effects consistent with other studies, suggesting some asthmatics susceptible. Evidence for mild airway inflammation.
Kim et al. (1991)	Seattle, Washington, USA	9 healthy athletes	Air, 0.18, and 0.30 ppm NO_2 for 30 min with exercise	No effects on pulmonary function. Symptoms not reported.	Small number of subjects limits conclusions.
Morrow et al. (1992)	Rochester, NY, USA	20 COPD subjects (14 current smokers) and 20 elderly healthy (13 never-smokers, 4 former smokers, 3 current smokers)	Air vs. 0.3 ppm NO_2 for 4 h with intermittent exercise.	COPD: small declines in FVC and FEV_1 with NO_2. Healthy: No symptoms or pulmonary function effects for group as a whole. Healthy smokers showed a 2.3% decline in FEV_1 with NO_2, and differed from nonsmokers.	Mild lung function effects of 0.3 ppm for 4 h in exercising patients with COPD. Small number of healthy smoking subjects limits conclusions regarding this group.
Pathmanathan et al. (2003)	United Kingdom, Sweden	12 healthy nonsmokers	Air vs. 2 ppm NO_2 for 4 h on 4 days, with intermittent exercise. Bronchoscopy and biopsy 1 h after exposure.	Epithelial expression of IL-5, IL-10, IL-13, and ICAM-1 increased following NO_2 exposure. No data on inflammatory cells in BAL fluid.	Supportive evidence for pro-allergic airway inflammation favoring following NO_2 exposure.
Posin et al. (1978)	Downey, CA, USA	10 healthy nonsmokers	3 daily exposures for 2.5 h. 1^{st} day: air; 2nd and 3rd days: 1 or 2 ppm NO_2. Intermittent exercise. Subsequent control series of 3 daily air exposures.	Reduced hemoglobin and hematocrit, and red blood cell acetyl cholinesterase.	Suggests red blood cell effects of NO_2 (see Frampton et al., 2002). Exposures not randomized.
Rasmussen et al. (1992)	Denmark	14 healthy nonsmokers	Air vs. 2.3 ppm NO_2 for 5 h	Small increases in FVC and FEV_1. Reduced lung permeability and blood glutathione peroxidase after exposure.	Only 1 week between exposures may have confounded results.
Rigas et al. (1997)		12 healthy nonsmokers	2 h of 0.36 ppm NO_2, 0.75 ppm NO_2, 0.36 ppm SO_2, or 0.36 ppm O_3. Boluses of O_3 every 30 min to measure O_3 absorption.	NO_2 and SO_2 increased O_3 absorption by increasing biochemical substrates.	Suggests breathing mixtures of NO_2 and O_3 would increase O_3 dose to airways.

TABLE AX5.1 (cont'd). CLINICAL STUDIES OF NO_2 EXPOSURE IN HEALTHY SUBJECTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Sandström et al. (1990)	Sweden	32 healthy nonsmokers, 4 groups of 8 subjects	4 ppm NO_2 for 20 min with 15 min exercise. BAL 4, 8, 24, 72 h after exposure, compared with non-exposure control BAL	Increase in BAL mast cells and lymphocytes 4-24 h after exposure.	Study weakened by lack of control air exposure.
Sandström et al. (1991)	Sweden	18 healthy nonsmokers	2.25, 4.0, 5.5 ppm NO_2 for 20 min with light exercise. BAL 24 h after exposure, compared with non-exposure control BAL	Increase in BAL mast cells (all concentrations) and lymphocytes (4.0 and 5.5 ppm).	Study weakened by lack of control air exposure.
Sandström et al. (1992a)	Sweden	10 healthy nonsmoking men	4 daily exposures to 4 ppm NO_2 for 20 min with 15 min exercise. BAL 24 h after exposure, compared with non-exposure control BAL.	Reduction in alveolar macrophages, NK cells, and CD8 lymphocytes in BAL; reduction in total lymphocytes in blood.	Study weakened by lack of control air exposure.
Sandström et al. (1992b)	Sweden	8 healthy nonsmokers	1.5 ppm NO_2 for 20 min with 15 min exercise, every 2nd day $\times$ 6. BAL 24 h after exposure compared with non-exposure control BAL.	Reduced CD8+ T lymphocytes and NK cells in BAL fluid.	Study weakened by lack of control air exposure.
Solomon et al. (2000)	San Francisco, California, USA	15 healthy nonsmokers	Air or 2.0 ppm NO_2 with intermittent exercise, for 4 h daily $\times$ 4. BAL 18 hours after exposure.	Increased neutrophils in bronchial lavage decreased CD4+ T lymphocytes in BAL. No changes in blood.	Airway inflammation with 2 ppm NO_2 for 4 daily 4 h exposures.
Vagaggini et al. (1996)	Italy	7 healthy nonsmokers	Air vs. 0.3 ppm NO_2 for 1 h with intermittent exercise.	Mild increase in symptoms. No effects on lung function, nasal lavage, or induced sputum.	Small number of subjects limits statistical power.

TABLE AX5.2. EFFECTS OF NO_2 EXPOSURE IN SUBJECTS WITH RESPIRATORY DISEASE (SEE TABLE AX5-3 FOR STUDIES WITH ALLERGEN CHALLENGE)

Reference	Location	Participants	Approach & Methods	Findings	Comments
Gong et al. (2005)	Downey, CA, USA	6 healthy nonsmokers and 18 ex-smokers with COPD	2 h exposures with intermittent exercise to: 1) air, 2) 0.4 ppm NO_2, 3) 200 µg/m^3 concentrated ambient particulate matter (CAPs), 4) NO_2 + CAPs	Reduced maximum mid-expiratory flow rate and oxygen saturation with CAPs exposures; no effects of NO_2 alone or additive effect with CAPs.	Exposures not fully randomized. Small number of subjects limits interpretation for healthy group.
Hackney et al. (1992)	Downey, CA, USA	26 smokers with symptoms and reduced FEV_1	Personal monitoring and chamber exposure to air and 0.3 ppm NO_2 for 4 h with intermittent exercise	No significant effects on lung function.	
Jörres and Magnussen (1991)	Germany	11 mild asthmatics	Air vs. 0.25 ppm NO_2 for 30 min with 10 min exercise	No effects on lung function or airways responsiveness to methacholine.	
Jörres et al. (1995)	Germany	8 healthy nonsmokers & 12 mild asthmatics	Air or 1 ppm NO_2 exposure for 3 h with intermittent exercise.	In asthmatics, 2.5% decrease FEV_1 after NO_2 vs. 1.3% decrease after air, $p = 0.01$. FEV_1 decreased 20% in 1 subject after NO_2. No significant lung function effect in healthy subjects. Changes in eicosanoids (more pronounced in asthmatics), but not inflammatory cells, in BAL fluid.	Lung function effects consistent with other studies, suggesting some asthmatics susceptible. Evidence for mild airway inflammation. Small number of healthy subjects limits statistical power.
Morrow et al. (1992)	Rochester, NY, USA	20 COPD, 20 healthy elderly	Air vs. 0.3 ppm NO_2 for 4 h with intermittent exercise	Equivocal reduction in FVC with COPD patients, but not healthy subjects.	
Strand et al. (1996)	Sweden	19 mild asthmatics	Air vs. 0.26 ppm NO_2 for 30 min with intermittent exercise	Increased airway responsiveness to histamine 5 h after exposure. No effects on lung function.	Suggests increased nonspecific airways responsiveness at much lower concentration than healthy subjects. Differs from findings in Jörres and Magnussen 1991
Vagaggini et al. (1996)	Italy	8 mild asthmatics, 7 COPD	Air vs. 0.3 ppm NO_2 for 1 h with intermittent exercise.	Mild decrease in FEV_1 in COPD subjects in comparison with air exposure, but not with baseline. No effects on nasal lavage or induced sputum.	No convincing effect of NO_2 in this study. Small number of subjects limits statistical power.

TABLE AX5.3. EFFECTS OF NO_2 EXPOSURE ON RESPONSE TO INHALED ALLERGEN

Reference	Location	Participants	Approach & Methods	Findings	Comments
Barck et al. (2002)	Sweden	13 mild asthmatics, 4 ex-smokers	30 min exposures to air and 0.26 ppm NO_2 (at rest?), allergen challenge 4 h and BAL 19 h after exposure. Randomized, crossover, double blind.	Increased PMN in bronchial wash and BAL fluid, increased eosinophil cationic protein in bronchial wash, and reduced cell viability and BAL volume with NO_2 + allergen. No effects on lung function response to allergen.	Key study suggesting that NO_2 enhances inflammatory response to allergen in mild asthmatics.
Barck et al. (2005a)	Sweden	18 mild asthmatics, 4 ex-smokers	Day 1: 15 min exposures, Day 2: 2 15-min exposures to air and 0.26 ppm NO_2 separated by 1 h, at rest. Allergen challenge 4 h after exposure on day 1 and 3 h after exposure on day 2. Sputum induction before exposure on days 1 & 2, and morning of day 3. Randomized, crossover, single blind.	Increased eosinophilic cationic protein in sputum and blood, and increased myeloperoxidase in blood with NO_2 + allergen. No differences in lung function or sputum cells.	Provides supporting evidence that NO_2 enhances the airway inflammatory response to allergen.
Barck et al. (2005b)	Sweden	16 mild asthmatics with rhinitis	30 min exposures to air and 0.26 ppm NO_2 at rest, nasal allergen challenge 4 h after exposure. Nasal lavage before and at intervals after exposure and challenge.	No significant differences between air and NO_2 exposure.	0.26 ppm NO_2 did not enhance nasal inflammatory response to allergen challenge.
Devalia et al. (1994)	United Kingdom	8 mild asthmatics	6 h exposures to combination of 0.4 ppm NO_2 and 0.2 ppm SO_2.	Increased allergen responsiveness 10 min after exposure to combination of NO_2 and SO_2, but not to individual gases.	Small number of subjects limits statistical power.
Jenkins et al. (1999)	United Kingdom	11 mild asthmatics	1) 6-h exposures to air, 0.1 ppm ozone, 0.2 ppm NO_2, and combination followed by allergen challenge; 2) 3-h exposures to air, 0.2 ppm ozone, 0.4 ppm NO_2, and combination; All exposures with intermittent exercise.	All of the second exposure scenarios (ozone, NO_2, and combination), but none of the first exposure scenarios, resulted in reduced concentration of allergen causing a 20% decline in FEV_1. Authors conclude that concentration more important than total inhaled pollutant.	Suggests 0.4 ppm for 3 h with intermittent exercise increases allergen responsiveness.

TABLE AX5.3 (cont'd). EFFECTS OF NO_2 EXPOSURE ON RESPONSE TO INHALED ALLERGEN

Reference	Location	Participants	Approach & Methods	Findings	Comments
Rusznak et al. (1996)	United Kingdom	13 mild asthmatics	6 h exposures to combination of 0.4 ppm NO_2 and 0.2 ppm SO_2	Increased allergen responsiveness to combination of NO_2 and SO_2, 10 min, 24, and 48 h after exposure.	Confirms findings of Devalia et al. (1994), that NO_2 + SO_2 for 6 h increases allergen responsiveness.
Strand et al. (1997)	Sweden	18 patients with mild asthma, age 18-50 yrs	Exposure to 0.26 ppm NO_2 for 30 min at rest, allergen challenge 4 h after exposure	Late phase, but not early phase, response to allergen enhanced by NO_2.	Suggests 0.26 ppm NO_2 for 30 min at rest increases late response.
Strand et al. (1998)	Sweden	16 patients with mild to moderate asthma, age 21-52 yrs	4 daily repeated exposures to 0.26 ppm NO_2 for 30 min at rest	Significant increases in both early and late phase response to allergen after 4th day of exposure.	Suggests repeated 0.26 ppm NO_2 at rest increases allergen response.
Tunnicliffe et al. (1994)	United Kingdom	10 nonsmoking mild asthmatics age 16-60 yrs. 8 subjects completed.	Exposure to air, 0.1 ppm, and 0.4 ppm NO_2 for 1 h at rest, separated by at least 1 week, followed by allergen challenge	Post-challenge reduction in FEV_1 after 0.4 ppm NO_2 was greater than after air, for both the early ($p < 0.009$) and late ($p < 0.02$) responses. No difference in nonspecific airway responsiveness.	Suggests threshold for allergen responsiveness effect is between 0.1 and 0.4 ppm for 1 h resting exposure.
Wang et al. (1995a); Wang et al. (1995b)	United Kingdom	2 groups of 8 subjects with allergic rhinitis	Exposure to 0.4 ppm NO_2 (at rest?) for 6 h followed by nasal allergen challenge and nasal lavage	Increase in myeloperoxidase and eosinophil cationic protein in nasal lavage fluid following allergen challenge.	Suggests enhanced nasal inflammatory response to allergen with 0.4 ppm.
Wang et al. (1999)	United Kingdom	16 subjects with allergic rhinitis	Treatment with nasal fluticasone or placebo for 4 weeks followed by exposure to 0.4 ppm NO_2 for 6 h, allergen challenge, and nasal lavage	Fluticasone suppressed the NO_2 and allergen–induced increase in eosinophil cationic protein in nasal lavage fluid.	Confirms earlier findings of this group that 0.4 ppm NO_2 enhances nasal allergen response.

TABLE AX5.4. EFFECTS OF EXPOSURE TO NO_2 WITH OTHER POLLUTANTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Devalia et al. (1994)	United Kingdom	8 mild asthmatics	6 h exposures to combination of 0.4 ppm NO_2 and 0.2 ppm SO_2.	Increased allergen responsiveness 10 min after exposure to combination of NO_2 and SO_2, but not to individual gases.	Small number of subjects limits statistical power.
Drechsler-Parks (1995)	Santa Barbara, CA, USA	8 older healthy nonsmokers	4 2-h exposures with intermittent exercise: air, 0.60 ppm NO_2, 0.45 ppm O_3, and 0.60 ppm NO_2 + 0.45 ppm O_3.	Significant reduction in cardiac output during exercise, estimated using noninvasive impedance cardiography, with NO_2 + O_3. Symptoms and pulmonary function not reported.	Suggests cardiac effects of NO_2 + O_3. Small number of subjects limits statistical power, has not been replicated.
Gong et al. (2005)	Downey, CA, USA	6 healthy nonsmokers and 18 ex-smokers with COPD	2 h exposures with intermittent exercise to: 1) air, 2) 0.4 ppm NO_2, 3) 200 $\mu g/m^3$ concentrated ambient particulate matter (CAPs), 4) NO_2 + CAPs	Reduced maximum mid-expiratory flow rate and oxygen saturation with CAPs exposures; no effects of NO_2 alone or additive effect with CAPs.	Exposures not fully randomized. Small number of healthy subjects limits interpretation for healthy group.
Hazucha et al. (1994)	Chapel Hill, North Carolina, USA	21 healthy female nonsmokers	2 h exposure to air or 0.6 ppm NO_2 followed 3 h later by exposure to 0.3 ppm O_3, with intermittent exercise.	NO_2 enhanced spirometric responses and airways responsiveness following subsequent O_3 exposure.	0.6 ppm NO_2 enhanced ozone responses.
Jörres and Magnussen (1990)	Germany	14 nonsmoking mild asthmatics	30 min exposures to air, 0.25 ppm NO_2, or 0.5 ppm SO_2 at rest followed 15 min later by 0.75 ppm SO_2 hyperventilation challenge.	NO_2 but not SO_2 increased airways responsiveness to SO_2 challenge.	Findings contrast with Rubenstein, et al. (1990).
Koenig et al. (1994)	Seattle, Washington, USA	28 asthmatic adolescents; 6 subjects did not complete.	Exposure for 90 min with intermittent exercise to: 1) 0.12 ppm ozone + 0.3 ppm NO_2, 2) 0.12 ppm ozone + 0.3 ppm NO_2 + 68 $\mu g/m^3$ H_2SO_4, or 3) 0.12 ppm ozone + 0.3 ppm NO_2 + 0.05 ppm nitric acid.	No effects on pulmonary function	Absence of lung function effects of 0.3 ppm NO_2 consistent with other studies; no effects of mixtures.
Rubenstein et al. (1990)	San Francisco, California, USA	9 stable asthmatics	30 min exposures to air or 0.3 ppm NO_2 with 20 min exercise, followed 1 h later by SO_2 inhalation challenge.	No effects on pulmonary function or SO_2 responsiveness.	Findings contrast with Jörres & Magnussen et al. (1990).

TABLE AX5.4 (cont'd). EFFECTS OF EXPOSURE TO NO_2 WITH OTHER POLLUTANTS

Reference	Location	Participants	Approach & Methods	Findings	Comments
Rudell et al. (1999)	Sweden	10 healthy nonsmokers	Air and diesel exhaust for 1 h, with and without particle trap. NO_2 concentration 1.2-1.3 ppm. BAL 24 h after exposures.	Increased neutrophils in BAL fluid, no significant reduction in effect with particle trap.	Filter only partially trapped particles. Unable to draw conclusions about role of NO_2 in causing effects.
Rusznak et al. (1996)	United Kingdom	13 mild asthmatics	6 h exposures to combination of 0.4 ppm NO_2 and 0.2 ppm SO_2.	Increased allergen responsiveness to combination of NO_2 and SO_2, 10 min, 24, and 48 h after exposure.	Confirms findings of Devalia et al. (1994), that NO_2 + SO_2 for 6 h increases allergen responsiveness.

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AX6. CHAPTER 6 ANNEX – EPIDEMIOLOGICAL STUDIES OF HUMAN HEALTH EFFECTS ASSOCIATED WITH AMBIENT OXIDES OF NITROGEN EXPOSURE

TABLE AX6.1. STUDIES EXAMINING EXPOSURE TO INDOOR NO_2 AND RESPIRATORY SYMPTOMS

Author, Year/Outcome	OR or RR (95% CI)	Subjects/Location	Analysis/Monitoring Device	NO_2 Measurement		
				Exposure Time	Mid-Range (ppb)	Range (ppb)
Pilotto et al. (2004)		118 asthmatic children/Australia	negative binomial/passive diffusion badges	6 h	mean (sd) intervention 16 (7)	7, 38
daytime symptoms					mean (sd) control 47 (27)	12, 116
difficulty breathing	RR 2.44 (1.02, 14.29)*					
chest tightness	RR 2.22 (1.23, 4.00)*					
asthma attacks	RR 2.56 (1.08, 5.88)*					
difficulty breathing, night	RR 3.12 (1.45, 7.14)*					
Pilotto et al. (1997)	OR 1.41 (0.63, 3.15)	388 children/Australia	generalized linear mixed models/passive diffusion badges	6 h		4, 132
wheeze (>40 ppb)						
Nitschke et al. (2006)		174 asthmatic children/Australia	negative binomial/passive diffusion badges	6 h	mean home 20 (22)	
night symptoms					mean school 34 (28)	
difficulty breathing						
school max	RR 1.23 (1.10, 1.39)					
home max	RR 1.06 (1.02, 1.10)					
chest tightness						
school max	RR 1.25 (1.14, 1.37)					

TABLE AX6.1 (cont'd). STUDIES EXAMINING EXPOSURE TO INDOOR NO_2 AND RESPIRATORY SYMPTOMS

Author, Year	OR or RR (95% CI)	Subjects/Location	NO_2 Measurement: Analysis/Monitoring Device	Exposure Time	Mid-range (ppb)	Range (ppb)
Garrett et al. (1998)		148 children/Australia	multiple logistic regression/passive monitors	4 days	med 6	p10-p90, 3, 15
chest tightness	OR 1.53 (0.45, 5.32)					
Smith et al. (2000)		125 asthmatic adults/children/Australia	GEE/passive diffusion badges	4.5 h		4, 147
children (n = 49, 0-14)						
chest tightness	OR 1.12 (1.07, 1.18)					
Belanger et al. (2006)		728 asthmatic children/Northeast US	logistic, Poisson regression/Palmes tubes	2 wks	mean (sd) gas home 26 (18)	
multifamily housing					mean (sd) elect home 9 (9)	
wheeze	RR 1.33 (1.05, 1.68)					
chest tightness	RR 1.51 (1.18, 1.91)					
Chauhan et al. (2003)		114 asthmatic children/ Southampton U.K.	Palmes diffusion tubes	7 d	Exposure tertiles: < 4; 4-7 ; > 7	Chauhan et al. (2003)
Increased symptom score, comparing first and second tertiles of NO_2 exposure	0.6 (0.01, 1.18)					
Increased symptom score, comparing first and third tertiles of NO_2 exposure	2.1 (0.52, 3.81)					

TABLE AX6.1 (cont'd). STUDIES EXAMINING EXPOSURE TO INDOOR NO_2 AND RESPIRATORY SYMPTOMS

Author, Year	OR or RR (95% CI)	Subjects/Location	NO_2 Measurement: Analysis/Monitoring Device	Exposure Time	Mid-range (ppb)	Range (ppb)
van Strien et al. (2004)		762 infants/Northeast US	Poisson regression		med 10	
persistent cough						
<5.1 ppb	RR 1.0					
5.1, 9.9 ppb	RR 0.96 (0.69, 1.36)					
9.9, 17.4 ppb	RR 1.33 (0.94, 1.88)					
>17.4 ppb	RR 1.52 (1.00, 2.31)					
shortness of breath						
<5.1 ppb	RR 1.0					
5.1, 9.9 ppb	RR 1.59 (0.96, 2.62)					
9.9, 17.4 ppb	RR 1.95 (1.17, 3.27)					
>17.4 ppb	RR 2.38 (1.31, 4.34)					

Notes:
Unless otherwise noted, results given for 20 ppb increase in NO_2.
*For purpose of comparison, RRs from Pilotto et al. (2004) are shown here as risk of symptoms given greater exposure to NO_2, i.e., control (unflued gas heater) vs. intervention (flued or electric replacement heater).
RRs reported by Pilotto el al. (2004) as protective effects for intervention vs. control.

TABLE AX6.2. STUDIES EXAMINING EXPOSURE TO AMBIENT NO_2 AND ACUTE RESPIRATORY SYMPTOMS USING GENERALIZED ESTIMATING EQUATIONS (GEE) IN THE ANALYSIS METHOD

				NO_2 Measurement			Correlation with Other Pollutants				
Author, Year	**OR (95% CI)**	**Location**	**Subjects**	**Avg Time**	**Mid-range (ppb)**	**Range (ppb)**	**$PM_{2.5}$**	**PM_{10}**	**O_3**	**SO_2**	**CO**
Children: Multicity Studies											
Schwartz et al. (1994)		US, 6-Cities	1844 children	24 h	med 13	p10-p90, 5, 24	0.35	0.36	−0.28	0.51	
cough, incidence: lag 1-4 mean	1.61 (1.08, 2.43)										
Mortimer et al. (2002)		US, NCICAS	864 asthmatic children	4 h	med 25	7, 90			0.27		
asthma symptoms: lag 1-6 mean	1.48 (1.02, 2.16)										
Schildcrout et al. (2006)		North America, CAMP	990 asthmatic children	24 h	med 23	min p10 to max p90, 10, 37		0.26, 0.64	0.04, 0.47	0.23, 0.68	0.63, 0.92
asthma symptoms: lag 0	1.06 (1.00, 1.13)										
lag 1	1.04 (0.97, 1.10)										
lag 2	1.09 (1.03, 1.15)										
3-day moving sum	1.04 (1.01, 1.07)										

TABLE AX6.2 (cont'd). STUDIES EXAMINING EXPOSURE TO AMBIENT NO_2 AND ACUTE RESPIRATORY SYMPTOMS USING GENERALIZED ESTIMATING EQUATIONS (GEE) IN THE ANALYSIS METHOD

				NO_2 Measurement			Correlation with Other Pollutants				
Author, Year	**OR (95% CI)**	**Location**	**Subjects**	**Avg time**	**Mid-range (ppb)**	**Range (ppb)**	**$PM_{2.5}$**	**PM_{10}**	**O_3**	**SO_2**	**CO**
Children: Single City Studies											
Pino et al. (2004)		Chile	504 infants	24 h	mean (sd) 41 (19)	p5-p95, 20, 81					
wheezy bronchitis: 6-day lag	1.14 (1.04, 1.30)										
Ostro et al. (2001)		Southern CA	138 asthmatic children, African American	1 h	mean (sd) 80 (4)	20, 220	0.34	0.63	0.48		
cough, incidence: lag 3	1.07 (1.00, 1.14)										
wheeze, incidence: lag 3	1.05 (1.01, 1.09)										
Delfino et al. (2002)		Southern CA	22 asthmatic children	8 h	mean (sd) 15 (7)	6, 34		0.55	0.26		
asthma symptoms: lag 0	1.91 (1.07, 3.39)										
Segala et al. (1998)		Paris	84 asthmatic children	24 h	mean (sd) 30 (8)	13, 65	(0.61)*	0.55		0.54	
asthma symptoms, incidence: lag 0	1.89 (1.13, 3.17)										
lag 1	1.36 (0.70, 2.64)										
lag 4	1.80 (1.07, 3.01)										
nocturnal cough, incidence: lag 3	1.44 (0.99, 2.08)										
lag 4	1.74 (1.20, 2.52)										

TABLE AX6.2 (cont'd). STUDIES EXAMINING EXPOSURE TO AMBIENT NO_2 AND ACUTE RESPIRATORY SYMPTOMS USING GENERALIZED ESTIMATING EQUATIONS (GEE) IN THE ANALYSIS METHOD

				NO_2 Measurement			Correlation with Other Pollutants				
Author, Year	OR (95% CI)	Location	Subjects	Avg time	Mid-range (ppb)	Range (ppb)	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO
Just et al. (2002)		Paris	82 asthmatic children	24 h	mean (sd) 29 (9)	12, 59	0.92*	0.54	0.09	0.69	
nocturnal cough, incidence: lag 0	2.11 (1.20, 3.74)										
lag 0-2	1.80 (0.89, 3.84)										
lag 0-4	1.58 (0.73, 3.54)										
Jalaludin et al. (2004)		Australia	148 children with wheeze history	15 h	mean (sd) 15 (6)	3, 79		0.26	−0.31		
wet cough: lag 0	1.13 (1.00, 1.26)										
Pino et al. (2004)											
Adults											
Segala et al. (2004)		Paris	46 nonsmoking adults	24 h	mean (sd) 30 (9)	12, 71	0.82*	0.83			
sore throat, cough: lag 0-4	4.05 (1.20, 13.60)										
von Klot et al. (2002)		Germany	53 asthmatic adults	24 h	med 24	4, 63		0.74		0.36	0.82
wheeze, prev: 5-day mean	1.15 (1.02, 1.31)										
phlegm, prev: 5-day mean	1.22 (1.10, 1.39)										
cough, prev: 5-day mean	1.15 (1.00, 1.31)										
breathing prob in a.m.: 5-day mean	1.25 (1.10, 1.39)										

Odds ratios (OR) given for 20 ppb increase in NO_2 with 24-h averaging time, or 30 ppb for 1-h averaging time. (20 ppb increases also used for times between 1 and 24 h.) *BS

TABLE AX6.3-1. RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
UNITED STATES				
Moolgavkar (2000a,b,c) Moolgavkar (2003) Multi-city, United States: Chicago, Los Angeles, Maricopa County, (Phoenix). Period of Study: 1987-1995	Outcomes (ICD 9 codes): COPD including asthma (490-496) Age groups analyzed: 0-19, 20-64, 65+ (LA only) Study Design: Time series Statistical Analyses: Poisson regression, GAM Covariates: day of wk, temporal trends, temperature, relative humidity Lag: 0-5 days	Chicago Median: 25 ppb IQR: 10 ppb Los Angeles Median: 38 ppb IQR: 18 ppb Maricopa Median: 19 ppb IQR: 12 ppb	Chicago: PM_{10}; r = 0.49 CO; r = 0.63 SO_2; r = 0.44 O_3; r = 0.02 LA: $PM_{2.5}$; r = 0.73 PM_{10}; r = 0.70 CO; r = 0.80 SO_2; r = 0.74 03; r = −0.10 Maricopa: PM_{10}; r = 0.22 CO; r = 0.66 SO_2; r = 0.02 O_3; r = −0.23	Increment: 10 ppb COPD, >65 yrs Chicago 1.7% [CI 0.36, 3.05] lag 0 - GAM default Chicago 2.04% [t = 2.99] lag 0 - GAM-100 Los Angeles 2.5% [CI 1.85, 3.15] lag 0 - GAM default Los Angeles 2.84% [t = 13.32] lag 0 - GAM - 30 Los Angeles 1.80% [t = 9.60] lag 0 - GAM - 100 Los Angeles 1.78% [t = 7.72] lag 0 - NS-100 Phoenix 4.4% [CI 1.07, 7.84] lag 5 Chronic Respiratory Disease Los Angeles 0-19 yrs 4.9% [CI 4.1, 5.7] lag 2 20-64 yrs 1.7% [CI 0.9, 2.1] lag 2 Multi-pollutant model NO_2 and PM_{10}: 1.72% [t = 3.18] lag 0 - GAM-100 NO_2 and $PM_{2.5}$: 1.51% [t = 2.07] lag 0 - GAM-100
Moolgavkar* et al. (1997) United States: Minneapolis-St. Paul Period of Study: 1986-1991	Outcomes (ICD 9 codes): COPD including asthma (490-496), Pneumonia (480-487) Age groups analyzed: 65+ Study Design: Time series Statistical Analyses: Semi-parametric Poisson regression, GAM Covariates: day of wk, season, temporal trends, temperature Statistical Package: S Plus Lag: 0-3 days	NO_2 24-h avg (ppb) 16.3 ppb IQR: 9.5 ppb	PM_{10}; r = 0.31 SO_2; r = 0.09 CO; r = 0.58	Increment: 10 ppb Sum of Pneumonia and COPD 2.2% [0.2, 4.2] lag 1 Pneumonia Only 3.1% [0.6, 5.6] lag 1, 20 df 1.7% [-0.8, 4.2] lag 1, 130 df

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Neidell (2004) California Period of Study: 1992-1998	Outcomes (ICD 9 codes): Asthma Age groups analyzed: <18; 0-1; 1-3; 3-6; 6-12; 12-18 Study Design: Time series Statistical Analyses: NR Covariates: Temperature, precipitation, influenza epidemic, Seasons: Nov-Mar only Lag: 0-4 days	NO_2 (ppb) Mean: 45.947 SD: 17.171	O_3 CO PM_{10}	Increment: NR Age 0-1 Fixed effects: 0.009 (0.014) Controlled for avoidance behavior: 0.009 (0.014) Single pollutant: 0.001 (0.011) Adjusted for SES: 0.021 (0.017) Interaction with Low SES: −0.017 (0.029) Age 1-3 Fixed effects: 0.002 (0.016) Controlled for avoidance behavior: 0.002 (0.016) Single pollutant: 0.009 (0.013) Adjusted for SES: −0.001 (0.020) Interaction with Low SES: −0.004 (0.032) Age 3-6 Fixed effects: 0.006 (0.016) Controlled for avoidance behavior: 0.006 (0.016) Single pollutant: 0.028 (0.013) Adjusted for SES: 0.020 (0.020) Interaction with Low SES: −0.037 (0.033) Age 6-12 Fixed effects: 0.041 (0.015) Controlled for avoidance behavior: 0.042 (0.015) Single pollutant: 0.047 (0.012) Adjusted for SES: 0.040 (0.018) Interaction with Low SES: −0.016 (0.031) Age 12-18 Fixed effects: 0.005 (0.013) Controlled for avoidance behavior: 0.005 (0.013) Single pollutant: 0.015 (0.010) Adjusted for SES: 0.013 (0.017) Interaction with Low SES: −0.020 (0.026)

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Karr et al. (2006) Southern LA County, CA, United States Period of Study: 1995-2000	Outcomes (ICD 9 codes): Acute bronchiolitis (466.1) Age groups analyzed: 0-1 yr Study Design: Case-crossover N: 19,109 Statistical Analyses: Conditional logistic regression Covariates: day of wk, temperature, humidity Seasons: Nov-Mar only Lag: 0-4 days	1-h max NO_2 (ppb) Mean: 59 ppb IQR: 26 ppb Number of Stations: 34	CO $PM_{2.5}$	Increment: 26 ppb (IQR) Acute bronchiolitis OR 0.96 [0.94, 0.99] lag 4 OR 0.97 [0.95, 0.99] lag 1 Stratified by Gestational Age at Birth: 37-44 wks 0.98 [0.95, 1.00] lag 1; 0.97 [0.94, 0.99] lag 4 34-36 wks 0.90 [0.84, 0.97] lag 1; 0.94 [0.88, 1.02] lag 4 29-33 wks 1.01 [0.91, 1.13] lag 1; 0.90 [0.80, 1.01] lag 4 25-28 wks 0.94 [0.78, 1.13] lag 1; 0.90 [0.73, 1.11] lag 4

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Linn et al. (2000) Los Angeles, United States Period of Study: 1992-1995	Outcomes (ICD 9 codes): Asthma (493), COPD (APR-DRG 88), Pulmonary diagnoses (APR-DRG 75-101) Age groups analyzed: >30 Study Design: Time series N: 302,600 Statistical Analyses: Poisson regression, GAM, OLS regression Covariates: day of wk, holiday, max temperature, min temperature, rain days, mean temperature, barometric pressure, season Seasons: Winter (Jan-Mar), Spring (Apr-Jun), Summer (Jul-Sep), Fall (Oct-Dec) Statistical Package: SPSS and SAS Lag: 0, 1 days	All concentrations are in ppb. Winter: 3.4 ± 1.3 Spring: 2.8 ± 0.9 Summer: 3.4 ± 1.0 Autumn: 4.1 ± 1.4 Overall: 3.4 ± 1.3	Winter: CO; r = 0.89 PM_{10}; r = 0.88 O_3; r = −0.23 Spring: CO; r = 0.92 PM_{10}; r = 0.67 O_3; r = 0.35 Summer CO; r = 0.94 PM_{10}; r = 0.80 O_3; r = 0.11 Winter CO; r = 0.84 PM_{10}; r = 0.80 O_3; r = −0.00	Increment: 10 ppb All pulmonary All seasons: 0.7% ± 0.3% Winter: 1.1% ± 0.5% Spring: 0.7% ± 0.1% Summer: 0.4% ± 0.8% Autumn: 1.2% ± 0.4% Asthma All season: 1.4% ± 0.5% Winter: 2.8% ± 0.1% Spring: NR Summer: NR Autumn: 1.9% ± 0.8% COPD All season: 0.8% ± 0.4% Winter: NR Spring: NR Summer: NR Autumn: 1.6% ± 0.6%

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Gwynn* et al. (2000) Buffalo, NY United States Period of Study: 1988-1990 Days: 1,090	Outcomes (ICD 9 codes): Respiratory admissions: Acute bronchitis/bronchiolitis (466); Pneumonia (480-4860); COPD and Asthma (490-493, 496) Age groups analyzed: 6 Study Design: Time series N: 24, Statistical Analyses: Poisson regression with GLM and GAM Covariates: season, day of wk, holiday, temperature, relative humidity Lag: 0-3 days	24-h avg NO_2 (ppb): Min: 4.0 25th: 15.5 Mean: 20.5 75th: 24.5 Max: 47.5	H^+ r = 0.22 SO_4^{2-} r = 0.36 PM_{10} r = 0.44 O_3 r = 0.06 SO_2 r = 0.36 CO r = 0.65 COH r = 0.72	Increment: 27 9 ppb (Max-Mean; IQR) NO_2 alone: Max-Mean RR 1.033 (t = 1.32) lag 1 IQR RR 1.01 (t = 1.32) lag 1
Zanobetti and Schwartz (2006) Boston, MA, United States Period of Study: 1995-1999	Outcomes (ICD 9 codes): Pneumonia (480-7) Age groups analyzed: 65+ Special Population: Medicare patients only Study Design: Case-crossover N: 24,857 Statistical Analyses: Conditional logisitic regression Covariates: apparent temperature, day of wk Seasons: Warm (Apr-Sep), Cool (Oct-Mar) Statistical Package: SAS Lag: 0, 1 days, 0-1 avg	NO_2 median 23.20 ppb; 90-10%: 20.41 ppb; For lag 0-1 2 day avg 90-10% = 16.8 ppb; IQR = 10.83 Number of Stations: 5	$PM_{2.5}$; r = 0.55 BC; r = 0.70 CO; r = 0.67 O_3; r = −0.14	Increment: 20.41 ppb (90-10%) Pneumonia -0.16% [−4.73, 4.42] lag 0 Increment: 16.78 ppb (90-10%) Pneumonia 2.26% [−2.55, 7.01] lag 0-1

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA				
Burnett et al. (1997a) 16 cities Canada Period of Study: 4/1981-12/1991 Days: 3,927	Outcomes (ICD 9 codes): All respiratory admissions (466, 480-6, 490-4, 496) Study Design: Time series N: 720,519 # of hospitals: 134 Statistical Analyses: random effects relative risk regression model Covariates: long-term trend, season, day of wk, hospital, Statistical Package: NR Lag: 0, 1, 2 day	1-h max NO_2 (ppb) Mean: 35.5 SD: 16.5 25th: 25 50th: 33 75th: 43 95th: 62 99th: 87	O_3 r = 0.20 CO SO_2 COH	Increment: 10 ppb Single pollutant NO_2 and respiratory admissions, p = 0.772 Multipollutant model (adjusted for CO, O_3, SO_2, COH, dew point): RR 0.999 [0.9922, 1.0059] lag 0
Yang et al. (2003) Vancouver, Canada Period of Study: 1986-1998 Days: 4748	Outcomes (ICD 9 codes): All respiratory admissions (460-519) Study Design: Case-crossover Age groups analyzed: <3, ≥65 Statistical Analyses: conditional logistic regression Statistical Package: NR Lag: 0-5 days	24-h avg NO_2 (ppb): Mean: 18.74 SD: 5.66 5th: 11.35 25th: 14.88 50th: 17.80 75th: 21.45 100th: 49.00 IQR: 5.57 Number of stations: 30	CO SO_2 O_3 r = −0.32 COH	Increment: 5.57 ppb (IQR) All Respiratory Admissions <3 yrs: NO_2 alone: OR 1.05 [1.02, 1.09] lag 1 $NO_2 + O_3$: OR 1.05 [1.02, 1.09] lag 1 $NO_2 + O_3 + CO + COH + SO_2$: OR 1.05 [0.99, 1.11] lag 1 All Respiratory Admissions ≥65 yrs: NO_2 alone: OR 1.05 [1.03, 1.07] lag 1 $NO_2 + O_3$: OR 1.04 [1.02, 1.07] lag 1 $NO_2 + O_3 + CO + COH + SO_2$: OR 1.05 [1.01, 1.08] lag 1

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Fung et al. (2006) Vancouver, BC, Canada Period of Study: 6/1/95-3/31/99	Outcomes (ICD 9 codes): All respiratory hospitalizations (460-519) Age groups analyzed: 65+ Study Design: (1) Time series, (2) Case-crossover, (3) DM-models (Dewanji and Moolgavkar 2000, 2002) N: 40,974 Statistical Analyses: (1) Poisson, (2) conditional logistic regression, (3) DM method – analyze recurrent data in which the occurrence of events at the individual level over time is available Covariates: day of wk Statistical Package: S-Plus and R Lag: Current day, 3 and 5 day lag	NO_2 24-h avg: Mean: 16.83 ppb, SD = 4.34; IQR: 5.43 ppb; range: 7.22, 33.89	CO; r = 0.74 COH; r = 0.72 SO_2; r = 0.57 PM_{10}; r = 0.54 $PM_{2.5}$; r = 0.35 $PM_{10-2.5}$; r = 0.52 O_3; r = −0.32	Increment: 5.43 ppb. (IQR) NO_2 Time series RR 1.018 [1.003, 1.034] lag 0 RR 1.024 [1.004, 1.044] lag 0-3 RR 1.025 [1.000, 1.050] lag 0-5 RR 1.027 [0.998, 1.058] lag 0-7 NO_2 Case-crossover RR 1.028 [1.010, 1.047] lag 0 RR 1.035 [1.012, 1.059] lag 0-3 RR 1.032 [1.006, 1.060] lag 0-5 RR 1.028 [0.997, 1.060] lag 0-7 NO_2 DM model RR 1.012 [0.997, 1.027] lag 0 RR 1.018 [1.000, 1.037] lag 0-3 RR 1.007 [0.988, 1.026] lag 0-5 RR 1.002 [0.981, 1.023] lag 0-7 DM method produced slightly higher RR estimates on O_3, SO_2, and $PM_{2.5}$ compared to time series and case-crossover, and slightly lower RR estimates on COH, NO_2, and PM_{10}, though the results were not significantly different from one another.

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Yang (2005) Vancouver, BC, Canada Period of Study: 1994-1998 Days: 1826	Outcomes (ICD 9 codes): COPD excluding asthma (490-2, 494, 496) Age groups analyzed: 65+ Study Design: Time series N: 6,027 Statistical Analyses: Poisson regression with GAM (with more stringent criteria) Covariates: temperature, relative humidity, day of wk, temporal trends, season Statistical Package: S-Plus Lag: 0-6 days, moving avgs	24-h avg: 17.03 ppb, SD = 4.48; IQR: 5.47 ppb; Range: 4.28, 33.89 Winter: 19.20 (4.86) Spring: 15.36 (3.72) Summer: 16.33 (4.57) Fall: 17.27 (3.77) Number of Stations: 31	PM_{10}; r = 0.61 SO_2; r = 0.61 CO; r = 0.73 O_3; r = −0.10	Increment: 5.5 ppb (IQR) COPD >65 yrs, year round RR 1.05 [1.01, 1.09] lag 0 RR 1.04 [1.00, 1.10] lag 0-1 RR 1.07 [1.01, 1.13] lag 0-2 RR 1.08 [1.02, 1.15] lag 0-3 RR 1.10 [1.03, 1.17] lag 0-4 RR 1.11 [1.04, 1.19] lag 0-5 RR 1.11 [1.04, 1.20] lag 0-6 Two-pollutant model PM_{10}: RR 1.03 [0.90, 1.17] lag 0 CO: RR 1.07 [0.96, 1.20] lag 0-6 O_3: RR 1.12 [1.04, 1.20] lag 0-6 Multipollutant models NO_2, CO, SO_2, O_3, PM_{10}: RR 1.01 [0.88, 1.16] NO_2, CO, SO_2, O_3: RR 1.06 [0.95, 1.19] NO_2 was strongest predictor of hospital admission for COPD among all gaseous pollutants in single-pollutant models

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Lin* et al. (2004) Vancouver, BC Canada Period of Study: 1987-1991	Outcomes (ICD 9 codes): Asthma (493) Age groups analyzed: 6-12 Study Design: Time series N: 3,754 (2,331 male, 1,423 female) Statistical Analyses: Semi-parametric Poisson regression with GAM (with default and more stringent criteria) Covariates: Trend, day of wk, Statistical package: S-Plus Lag: Cumulative 1-7 day	24-h avg NO_2 (ppb) Mean: 18.65 SD: 5.59 Min: 4.28 25th: 14.82 50th: 17.75 75th: 21.36 Max: 45.36 Number of stations: 30	CO r = 0.73 SO_2 r = 0.67 O_3 r = −0.03 $PM_{2.5}$ r = 0.37 PM_{10} r = 0.55	Increment: 6.54 ppb (IQR) Boys 6-12 yrs by SES status: Low; High Lag 1 RR 1.13 [1.04, 1.23]; 1.04 [0.95, 1.14] Lag 2 RR 1.13 [1.02, 1.24]; 1.06 [0.95, 1.18] Lag 3 RR 1.14 [1.02, 1.27]; 1.06 [0.94, 1.19] Lag 4 RR 1.14 [1.02, 1.28]; 1.05 [0.92, 1.19] Lag 5 RR 1.12 [0.99, 1.27]; 1.10 [0.96, 1.26] Lag 6 RR 1.12 [0.98, 1.28]; 1.07 [0.93, 1.23] Lag 7 RR 1.11 [0.97, 1.28]; 1.09 [0.94, 1.27] Girls 6-12 yrs by SES status: Low; High Lag 1 RR 1.07 [0.96, 1.19]; 1.01 [0.90, 1.13] Lag 2 RR 1.03 [0.91, 1.17]; 0.98 [0.85, 1.12] Lag 3 RR 1.04 [0.91, 1.20]; 0.98 [0.84, 1.13] Lag 4 RR 1.11 [0.95, 1.29]; 1.01 [0.86, 1.19] Lag 5 RR 1.11 [0.94, 1.30]; 0.99 [0.83, 1.17] Lag 6 RR 1.08 [0.91, 1.28]; 1.03 [0.86, 1.24] Lag 7 RR 1.07 [0.90, 1.28]; 1.09 [0.90, 1.32] Multipollutant model (adjusted for SO_2) Boys, Low SES: 1.16 [1.06, 1.28] lag 1 1.18 [1.03, 1.34] lag 4 Results presented are default GAM, but authors state that use of natural cubic splines with a more stringent convergence rate produced similar results

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Lin et al. (2003) Toronto, ON Period of Study: 1981-1993	Outcomes (ICD 9 codes): Asthma (493) Age groups analyzed: 6-12 Study Design: Bi-directional case-crossover N: 7,319 Statistical Analyses: Conditional logistic regression Covariates: Daily maximum and minimum temperatures and avg relative humidity Lag: Cumulative lag of 1-7 days.	NO_2 24-h avg: 25.24 ppb, SD = 9.04; IQR: 11 ppb; Range: 3.00, 82.00 Number of Stations: 4	CO; r = 0.55 SO_2; r = 0.54 PM_{10}; r = 0.52 O_3; r = 0.03 $PM_{2.5}$; r = 0.50 PM_{10}-2.5; r = 0.38	Increment: 11 ppb. (IQR) Boys 6-12 yrs; Girls 6-12 yrs Lag 0: OR 1.04 [0.99, 1.10]; 0.99 [0.92, 1.06] Lag 0-1: OR 1.07 [1.00, 1.14]; 1.03 [0.94, 1.12] Lag 0-2: OR 1.09 [1.01, 1.17]; 1.07 [0.96, 1.18] Lag 0-3: OR 1.10 [1.01, 1.20]; 1.09 [0.97, 1.21] Lag 0-4: OR 1.10 [1.00, 1.20]; 1.14 [1.02, 1.28] Lag 0-5: OR 1.12 [1.01, 1.23]; 1.16 [1.02, 1.31] Lag 0-6: OR 1.11 [1.00, 1.24]; 1.16 [1.02, 1.32]
Burnett et al. (1997b) Toronto, Canada Period of Study: 1992-1994	Outcomes (ICD 9 codes): Respiratory tracheobronchitis (480-6), COPD (491-4, 496) Study Design: Time series Statistical Analyses: Poisson regression, GEE, GAM Covariates: Temperature, dew point temperature, long-term trend, season, influenza, day of wk Seasons: summers only Lag: 0,1,2,3,4 days	Mean NO_2: 38.5 ppb IQR NO_2: 5.75 ppb Range: 12, 81 Number of Stations: 6-11	PM_{10}; r = 0.61 CO; r = 0.25 H^+; r = 0.25 SO_4; r = 0.34 TP; r = 0.61 FP; r = 0.45 CP; r = 0.57 COH; r = 0.61 O_3; r = 0.07 SO_2; r = 0.46	Increment: 5.75 ppb (IQR) Respiratory - Percent increase 4.4% [CI 2.4, 6.4], lag 0 Copollutant and multipollutant models RR (t-statistic): NO_2, COH: 1.018 (1.36) NO_2, H^+: 1.037 (3.61) NO_2, SO_4: 1.033 (3.05) NO_2, PM_{10}: 1.039 (2.85) NO_2, $PM_{2.5}$: 1.037 (3.13) NO_2, $PM_{10-2.5}$: 1.037 (2.96) NO_2, O_3, SO_2: 1.028 (2.45) NO_2, O_3, SO_2, COH: 1.010 (0.71) NO_2, O_3, SO_2, H^+: 1.027 (2.39) NO_2, O_3, SO_2, SO_4: 1.027 (2.36) NO_2, O_3, SO_2, PM_{10}: 1.028 (1.77) NO_2, O_3, SO_2, $PM_{2.5}$: 1.028 (2.26) NO_2, O_3, SO_2, $PM_{10-2.5}$: 1.022 (1.71)

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Burnett et al. (1999) Metro Toronto, Canada Period of Study: 1980-1994	Outcomes (ICD 9 codes): Asthma (493); Obstructive lung disease (490-2, 496); Respiratory Infection (464, 466, 480-7, 494) Study Design: Time series Statistical Analyses: Poisson regression model with stepwise analysis Covariates: long-term trends, season, day of wk, daily maximum temperature, daily minimum temperature, daily avg dew point temperature, daily avg relative humidity Statistical Package: S-Plus, SAS Lag: 0,1,2 days, cumulative	24 h mean: 25.2 ppb, SD 9.1, CV = 36; IQR = 23 Number of stations: 4	COH; r = NR $PM_{2.5}$; r = 0.50 $PM_{10-2.5}$; r = 0.38 PM_{10}; r = 0.52 CO; r = 0.55 SO_2; r = 0.54 O_3; r = −0.03	Increment: 25.2 ppb (Mean) 7.72 excess daily admissions due to pollution of all sorts. 40.4% increase; or 3 excess daily admissions traced to NO_2. Single-pollutant model percent increase (t statistic) Asthma: 3.33% (2.37) lag 0 OLD 2.21% (1.07) lag 1 Respiratory infection: 6.89% (5.53), lag 2 Multipollutant model percent increase (SE) Respiratory infection: NO_2 alone: 4.64 (SE ≥3) $NO_2 + SO_2 + O_3 + PM_{2.5}$: 4.04 (SE ≥2) $NO_2 + SO_2 + O_3 + PM_{10-2.5}$: 4.56 (SE ≥3) $NO_2 + SO_2 + O_3 + PM_{10}$: 4.16 (SE ≥3) $NO_2 + O_3 + PM_{2.5}$: 4.44 (SE ≥2)
Burnett* et al. (2001) Toronto, Canada Period of Study: 1980-1994	Outcomes (ICD 9 codes): Croup (464.4), pneumonia (480-486), asthma (493), acute bronchitis/bronchiolitis (466) Age groups analyzed: <2 yrs Study Design: Time series Statistical Analyses: Poisson regression with GAM Covariates: temporal trend, day of wk, temperature, relative humidity Statistical Package: S-Plus Lag: 0-5 days	1-h max NO_2 (ppb) Mean: 44.1 CV: 33 5th: 25 25th: 35 50th: 42 75th: 52 95th: 70 99th: 86 100th: 146 Number of stations: 4	O_3 r = 0.52 SO_2 CO $PM_{2.5}$ $PM_{10-2.5}$	Increment: NR All respiratory admissions: Single-pollutant: Percent increase: 20.2 (t = 3.43) lag 0-1 Multipollutant (adjusted for O_3): Percent increase: 7.1 (t = 1.09) lag 0-1

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
CANADA (cont'd)				
Luginaah et al. (2005) Windsor, ON, Canada Period of Study: 4/1/95-12/31/00	Outcomes (ICD 9 codes): Respiratory admissions (460-519) Age groups analyzed: 0-14, 15-64, 65+, all ages Study Design: (1) Time series and (2) case-crossover N: 4,214 # of Hospitals: 4 Statistical Analyses: (1) Poisson regression, GAM with natural splines (stricter criteria), (2) conditional logistic regression with Cox proportional hazards model Covariates: Temperature, humidity, change in barometric pressure, day of wk Statistical Package: S-Plus Lag: 1,2,3 days	NO_2 mean 1-h max: 38.9 ppb, SD = 12.3; IQR: 16 Number of stations: 4	SO_2; r = 0.22 CO; r = 0.38 PM_{10}; r = 0.33 COH; r = 0.49 O_3; r = 0.26 TRS; r = 0.06	Increment: 16 ppb (IQR) Time series, females; males All ages, lag 1 1.035 [0.971, 1.104]; 0.944 [0.886, 1.006] 0-14 yrs, lag 2 1.114 [0.994, 1.248]; 0.955 [0.866, 1.004] 15-65 yr, lag 3 1.121 [0.978, 1.285], 1.012 [0.841, 1.216] 65+ yr, lag 1 1.020 [0.930, 1.119]; 0.9196 [0.832, 1.016] Case-crossover, females; males All ages, lag 1 1.078 [0.995, 1.168]; 0.957 [0.883, 1.036] 0-14 yrs, lag 2 1.189 [1.002, 1.411]; 0.933 [0.810, 1.074] 15-65 yr, lag 3 1.114 [0.915, 1.356]; 0.972 [0.744, 1.268] 65+ yr, lag 1 1.081 [0.964, 1.212]; 0.915 [0.810, 1.034]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
AUSTRALIA/NEW ZEALAND				
Simpson et al. (2005a) Multi-city study, Australia (Sydney, Melbourne, Brisbane, Perth) Period of Study: 1996-1999	Outcomes (ICD 9/ICD 10): All respiratory (460-519/J00-J99 excluding J95.4-J95.9, RO9.1, RO9.8), asthma (493/J45, J46, J44.8), COPD (490-492, 494-496/J40-J44, J47, J67), pneumonia with bronchitis (466, 480-486/J12-17, J18.0 j18.1 J18.8 J18.9 J20 J21) Age groups analyzed: 15-64 (asthma), 65+ (all respiratory, COPD, asthma, pneumonia with bronchitis) Study Design: Time series Statistical Analyses: Followed APHEA2 protocol: (1) Single city: (a) GAM with default and more stringent criteria, (b) GLM with default and more stringent criteria, (c) penalized spline models. (2) Multicity meta analysis: random effects meta-analysis Covariates: Temperature, relative humidity, day of wk, holiday, influenza epidemic, brushfire/controlled burn Statistical Package: S-Plus, R Lag: 0,1,2 days	Maxi 1 h NO_2 ppb (range) Brisbane: 24.1 (2.1, 63.3) Sydney: 23.7 (6.5, 59.4) Melbourne: 23.7 (4.4, 66.7) Perth: 16.3 (1.9, 41.0)	Brisbane: O_3; r = 0.15 BSP; r = 0.50 Melbourne: O_3; r = 0.30 BSP; r = 0.29 Sidney: O_3; r = 0.24 BSP; r = 0.54 Perth: O_3; r = 0.28 BSP; r = 0.62	Increment: Maxi 1 h NO_2 IQR Meta-analysis: Respiratory ≥65 yrs 1.0027 [1.0015, 1.0039] lag 0-1 COPD and Asthma >65 yrs 1.0020 [1.0003, 1.0037] lag 0-1 Pneumonia and Acute Bronchitis >65 yrs 1.0030 [1.0011, 1.0048] lag 0-1 Multipollutant Model Respiratory ≥65 yrs NO_2 Alone: 1.0027 [1.0015,1.0039] lag0-1 NO_2+BSP: 1.0023 [1.0009, 1.0038] lag 0-1 NO_2+O_3: 1.0028 [1.0016, 1.0040] lag 0-1 GAM results from S-Plus and R similar to one another, but different than results from GLM. GAM results from S-Plus presented.

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
AUSTRALIA/NEW ZEALAND (cont'd)				
Barnett et al. (2005) Multicity, Australia/New Zealand; (Auckland, Brisbane, Canberra, Christchurch, Melbourne, Perth, Sydney) Period of Study: 1998-2001	Outcomes (ICD 9/ICD 10): All respiratory (460-519/J00-J99 excluding J95.4-J95.9, RO9.1, RO9.8), asthma (493/J45, J46, J44.8), COPD (490-492, 494-496/J40-J44, J47, J67), pneumonia with bronchitis (466, 480-486/J12-17, J18.0 j18.1 J18.8 J18.9 J20 J21) Age groups analyzed: 0, 1-4, 5-14 Study Design: Case-crossover Statistical Analyses: Conditional logistic regression, random effects meta-analysis Covariates: Temperature, current-previous day temperature, relative humidity, pressure, extremes of hot and cold, day of wk, holiday, day after holiday Season: Cool, May-Oct; Warm, Nov-Apr Statistical Package: SAS Lag: 0-1 days	24-h avg (ppb) (range): Auckland 10.2 (1.7, 28.9) Brisbane 7.6 (1.4, 19.1) Canberra 7.0 (0, 22.5) Christchurch 7.1 (0.2, 24.5) Melbourne 11.7 (2, 29.5) Perth 9.0 (2, 23.3) Sydney 11.5 (2.5, 24.5) IQR: 5.1 ppb Daily 1h max (range): Auckland 19.1 (4.2, 86.3) Brisbane 17.3 (4, 44.1) Canberra 17.9 (0, 53.7) Christchurch 15.7 (1.2, 54.6) Melbourne 23.2 (4.4, 62.5) Perth 21.3 (4.4, 48) Sydney 22.6 (5.2, 51.4) IQR: 9.0 ppb	BS; r = 0.39, 0.63 $PM_{2.5}$; r = 0.34, 0.68 PM_{10}; r = 0.21, 0.57 CO; r = 0.53, 0.73 SO_2; r = 0.15, 0.58 O_3; r = -0.15, 0.28	Increment: 5.1 ppb (24 h) or per 9 ppb (1-h max). (IQR) 24-h avg NO_2 (5.1 ppb change) Pneumonia and acute bronchitis 0 yrs 3.2% [−1.8, 8.4] lag 0-1 1-4 yrs 4.8% [−1.0, 11.0] lag 0-1 5-14 yrs (sample size too small) Respiratory 0 yrs 3.1% [−1.0, 7.3] lag 0-1 1-4 yrs 2.4% [−0.8, 5.7] lag 0-1 5-14 yrs 5.8% [1.7, 10.1] lag 0-1 Asthma 0 yrs No analysis (poor diagnosis) 1-4 yrs 2.6% [−1.3, 6.6] lag 0-1 5-14 yrs 6.0% [0.2, 12.1] lag 0-1 1 h NO_2 maximum (9.0 ppb change) Pneumonia and acute bronchitis 0 yrs 2..8% [−1.8, 7.7] lag 0-1 1-4 yrs 4.1% [−2.4, 11.0] lag 0-1 5-14 yrs (sample size too small) Respiratory 0 yrs 2.2% [−1.6, 6.1] lag 0-1 1-4 yrs 2.8% [0.7, 4.9] lag 0-1 5-14 yrs 4.7% [1.6, 7.9] lag 0-1 Asthma 0 yrs No analysis (poor diagnosis) 1-4 yrs 2.5% [−0.2, 5.2] lag 0-1 5-14 yrs 2.6% [−2.2, 7.6] lag 0-1

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
AUSTRALIA/NEW ZEALAND (cont'd)				
Erbas et al. (2005) Melbourne, Australia Period of Study : 2000-2001	Outcomes (ICD 10): Asthma (J45, J46) Age groups analyzed: 1-15 Study Design: Time series N: 8,955 # of Hospitals: 6 Statistical Analyses: Poisson regression, GAM and GEE Covariates: Day of wk Dose-response investigated?: Yes Statistical Package: NR Lag: 0,1,2 days	1 hour mean NO_2: 16.80 ppb, SD = 8.61; range: 2.43, 63.00	PM_{10} O_3	Increment: 90th-10th percentile Inner Melbourne; increment = 25.54 ppb RR 0.83 [0.68, 0.98] lag 0 Western Melbourne; increment = 28.86 ppb RR 1.15 [1.03, 1.27] lag 2 Eastern Melbourne; increment = 17.67 ppb RR 1.07 [0.93, 1.22] lag 0 South/Southeastern; increment = 17.74 ppb RR 0.98 [0.79, 1.18] lag 1
Hinwood et al. (2006) Perth, Australia Period of Study: 1992-1998	Outcomes (ICD 9): COPD (490-496, excluding 493); Pneumonia (480-489.99); Asthma (493) Age groups analyzed: <15, 65+, all ages Study Design: Case-crossover, time-stratified Statistical Analyses: Conditional logistic regression Covariates: Temperature, change in temperature, maximum humidity, holiday, day of wk Statistical Package: NR Lag: 0,1,2,3 days or cumulative 0-2 and 0-3 days	24 h Mean [Std. Dev] (10th and 90th centile) All year 10.3 [5.0] (4.4, 17.1) Summer 9.6 [4.8] (4.3, 15.7) Winter 11.1 [5.1] (4.8, 18.0) Daily 1-h max Mean [Std. Dev] All year 24.8 [10.1] (13.3, 37.5) Summer 24.9 [8.9] (12.4, 39.2) Winter 24.7 [11.1] (14.4, 35.7) Number of stations: 3	O_3, r = −0.06 CO, r = 0.57 BS, r = 0.39 PM_{10} $PM_{2.5}$	Increment: 1 ppb (all values were estimated from the graphs) All respiratory NO_2 (24 hr) ≥65 yrs OR 1.005 [1.001, 1.011] lag 1 All ages OR: 1.002 [0.998, 1.004] lag 1 Pneumonia NO_2 (24 hr) ≥65 yrs OR 1.006 [0.999, 1.014] lag 1 All ages OR: 1.002 [0.998, 1.010] lag 1 COPD NO_2 (24 hr) ≥65 yrs OR 1.004 [0.990, 1.012] lag 2 All ages OR: 1.001 [0.995, 1.010] lag 2 Asthma NO_2 (24 hr) 0-14 yrs OR: 1.002 [0.998, 1.004] lag 0 ≥65 yrs OR 0.996 [0.988, 1.002] lag 0 All ages OR: 1.001 [0.999, 1.003] lag 0

TABLE AX6.3-1. (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
AUSTRALIA/NEW ZEALAND (cont'd)				
Morgan et al. (1998a) Sydney, Australia Period of Study: 1990-1994	Outcomes (ICD 9): COPD (490-492, 494, 496); Asthma (493) Age groups analyzed: 1-14, 15-64, 65+, all ages Study Design: Timeseries # of hospitals: 27 Statistical Analyses: APHEA protocol, Poisson regression, GEE Covariates: long-term trend, temperature, dew point, day of wk, holiday Statistical Package: SAS Lag: 0,1,2 days and cumulative	24 h daily mean: 15 ppb, SD = 6, range: 0, 52, IQR: 11, 90-10th percentile: 17 Mean daily 1-h max: 29 ppb, SD = 3, range: 0, 139, IQR: 15, 90-10th percentile: 29 # of stations: 3-14, r = 0.52	24-h avg NO_2: PM(24 h), r = 0.53 PM (1 h),r = 0.51 O_3, r = -0.9 1-h max NO_2: PM(24 h),r = 0.45 PM (1 h),r = 0.44 O_3, r = 0.13	Increment: 90-10th percentile 24-h avg (17 ppb) Asthma: 1-14 yrs 3.28% [−1.72, 8.54] lag 0 15-64 yrs 2.29% [−2.97, 7.83] lag 0 COPD: >65 yrs 4.30% [−0.75, 9.61] lag 1 Daily 1-h maximum (29 ppb) Asthma: 1-14 yrs 5.29% [1.07, 9.68] lag 0 15-64 yrs. 3.18% [−1.53, 8.11] lag 0 COPD: 65+ yrs. 4.60% [−0.17, 9.61] lag 1 Multipollutant model (29 ppb) Asthma: 1-14 yrs. 5.95% [1.11, 11.02] lag 0 COPD: 65+ yrs. 3.70% [−1.03, 8.66] lag 1
Petroeschevsky et al. (2001) Brisbane, Australia Period of Study: 1987-1994 Days: 2922	Outcomes (ICD 9): All respiratory (460-519); Asthma (493) Age groups analyzed: 0-4, 5-14, 15-64, 65+, all ages Study Design: Timeseries N: 33,710 (13,246 = asthma) Statistical Analyses: APHEA protocol, Poisson regression, GEE Covariates: Temperature, humidity, season, infectious disease, day of wk, holiday Season: Summer, Autumn, Winter, Spring, All year Dose-response investigated?: Yes Statistical Package: SAS Lag: Single: 1,2,3 day Cumulative: 0-2, 0-4	Mean (range) 24-h avg: Overall: 139 (12, 497) Summer: 97 (20, 331) Autumn: 129 (33, 319) Winter: 179 (12, 454) Spring: 153 (35, 497) Mean (range) 1-h max Overall: 282 (35, 1558) Summer: 206 (35, 580) Autumn: 256 (70, 585) Winter: 354 (35, 805) Spring: 321 (35, 1558) # of stations: 3, r = 0.43, 0.53	Bsp O_3 SO_2	Increment: 10 ppb Respiratory (1-h max): 0-4 yrs 1.015 [0.996, 1.035] lag 3 5-14 yrs 0.985 [0.950, 1.021] lag 0 All ages 0.989 [0.977, 1.002] lag 1 Respiratory (24-h avg): 15-64 yrs 1.027 [0.984, 1.071] lag 0 >65 yrs 0.903 [0.851, 0.959] lag 5 Asthma (1-h max): 0-4 yrs 0.975 [0.947, 1.004] lag0 5-64 yrs 0.983 [0.949, 1.018] lag 1 All ages 0.962 [0.936, 0.989] lag 0-2

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE				
Anderson et al. (1997) Multicity, Europe (Amsterdam, Barcelona, London, Paris, Rotterdam) Period of study: 1977-1989 for Amsterdam and Rotterdam 1986-1992 for Barcelona 1987-1991 for London 1980-1989 for Milan 1987-1992 for Paris	Outcomes (ICD 9): COPD - unspecified bronchitis (490), chronic bronchitis (491), emphysema (492), chronic airways obstruction (496) Study Design: Time series Statistical Analyses: APHEA protocol, Poisson regression, meta-analysis Covariates: trend, season, day of wk, holiday, influenza, temperature, humidity Season: Cool, Oct-Mar; Warm, Apr-Sep Statistical Package: NR Lag: 0,1,2 days and 0-3 cumulative	24 h all year avg: ($\mu g/m^3$) Amsterdam: 50 Barcelona: 53 London: 67 Paris: 42 Rotterdam: 52 1-h max Amsterdam: 75 Barcelona: 93 London: 67 Paris: 64 Rotterdam: 78	SO_2 BS TSP O_3	Increment: 50 $\mu g/m^3$ Meta-analytic results - Weighted mean values from 6 cities COPD-Warm season 24 h 1.03 [1.00, 1.06] lag 1 1 h 1.02 [1.00, 1.05] lag 1 COPD-Cool season 24 h 1.01 [0.99, 1.03] 1 h 1.02 [0.99, 1.05] COPD-All Year 24 hr 1.019 [1.002, 1.047] lag 1 24 hr 1.026 [1.004, 1.036] lag 0-3, cumulative 1 hr 1.013 [1.003, 1.022] lag 1 1 hr 1.014 [0.976, 1.054] lag 0-3, cumulative
Atkinson et al. (2001) Multicity, Europe (Barcelona, Birmingham, London, Milan, Netherlands, Paris, Rome, Stockholm) Period of study: 1998-1997	Outcomes (ICD 9): Asthma (493), COPD (490-496), All respiratory (460-519) Study Design: Timeseries Statistical Analyses: APHEA protocol, Poisson regression, meta-analysis Covariates: season, temperature, humidity, holiday, influenza Statistical Package: NR Lag: NR	1-h max of NO_2 ($\mu g/m^3$) Barcelona: 94.4 Birmingham: 75.8 London: 95.9 Milan: 147.0 Netherlands: 50.1 Paris: 87.2 Rome: 139.7 Stockholm: 35.6	SO_2, O_3, CO, BS PM_{10}; r = Barcelona: 0.48 B'gham: 0.68 London: 0.70 Milan: 0.72 Netherlands: 0.64 Paris: 0.44 Rome: 0.32 Stockholm: 0.30	Increment: 10 $\mu g/m^3$ for PM_{10}; change in NO_2 not described. Asthma, 0 to 14 yrs: For PM_{10}: 1.2% [0.2, 2.3] For $PM_{10} + NO_2$: 0.1 [−0.8, 1.0] Asthma, 15 to 64 yrs: For PM_{10}: 1.1% [0.3, 1.8] For $PM_{10} + NO_2$: 0.4 [−0.5, 1.3] COPD + Asthma, ≥65 yrs For PM_{10}: 1.0% [0.4, 1.5] For $PM_{10} + NO_2$: 0.8 [−0.6, 2.1] All Respiratory, ≥65 yrs of age For PM_{10}: 0.9% [0.6, 1.3] For $PM_{10} + NO_2$: 0.7 [−0.3, 1.7]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Sunyer et al. (1997) Multicity, Europe (Barcelona, Helsinki, Paris, London) Period of Study: 1986-1992	Outcomes (ICD 9): Asthma (493) Age groups analyzed: <15, 15-64 Study Design: Timeseries Statistical Analyses: APHEA protocol, Poisson regression, GEE; meta-analysis Covariates: Humidity, temperature, influenza, soybean, long-term trend, season, day of wk Season: Cool, Oct-Mar; Warm: Apr-Sep Statistical Package: NR Lag: 0,1,2,3 and cumulative 1-3	24 h median (range) ($\mu g/m^3$) Barcelona: 53 (5, 142) Helsinki: 35 (9, 78) London: 69 (27, 347) Paris: 42 (12, 157) # of stations: Barcelona: 3 London: 2 Paris: 4 Helsinki: 8	SO_2 black smoke O_3	Increment: 50 $\mu g/m^3$ of 24-h avg for all cities combined. Asthma 15-64 yrs 1.029 [1.003, 1.055] lag 0-1 1.038 [1.008-1.068] lag 0-3, cumulative <15 yrs 1.026 [1.006, 1.049] lag 2 1.037 [1.004, 1.067] lag 0-3, cumulative 1.080 [1.025, 1.140] – Winter only Two-pollutant models: NO_2/Black smoke 15-64 yrs 1.055 [1.005, 1.109] lag 0-1 15-64 yrs 1.088 [1.025, 1.155] cumulative 0-3 <15 yrs 1.036 [0.956, 1.122] NO_2/SO_2 <15 yrs 1.034 [0.988, 1.082]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Schouten et al. (1996) Multicity, The Netherlands (Amsterdam, Rotterdam) Period of Study: 04/01/77-09/30/89	Outcomes (ICD 9): All respiratory (460-519), COPD (490-2, 494, 496), Asthma (493) Age groups analyzed: 15-64, 65+, all ages Study Design: Time series Statistical Analyses: APHEA protocol, Poisson regression Covariates: Long-term trend, season, influenza, day of wk, holiday, temperature, humidity Season: Cool, Nov-Apr; Warm: May-Oct Statistical Package: NR Lag: 0,1,2 days; and cumulative 0-1 and 0-3 day lags	24-h avg NO_2 Amsterdam Mean/Med: 50/50 μg/m^3 Rotterdam Mean: 54/52 μg/m^3 Daily max 1 h Amsterdam Mean/Med: 75/75 μg/m^3 Rotterdam Mean/Med: 82/78 μg/m^3 # of stations: 1 per city	SO_2 BS O_3	Increment: 100 μg/m^3 increment. All respiratory, Amsterdam 24 h mean; 1-h max 15-64 yrs RR 0.890 [0.783, 1.012]; 0.894 [0.821, 0.973] lag 1 >65 yrs RR 1.023 [0.907, 1.154]; 0.996 [0.918, 1.080] lag 2 All respiratory, Rotterdam 24 h mean; 1-h max (1985-89) 15-64 yrs RR 0.965 [0.833, 1.118]; 1.036 [0.951, 1.129] lag 1 >65 yrs RR 1.172 [0.990, 1.387]; 1.073 [0.970, 1.186] lag 0 COPD, Amsterdam, 24 h mean, All ages RR 0.937 [0.818, 1.079] lag 1 Asthma Amsterdam, 24 h mean , All ages RR 1.062 [0.887, 1.271] lag 2 COPD, Rotterdam 24 h mean All ages RR 1.051 [0.903, 1.223] lag 2

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Ponce de Leon et al. (1996) London, England Period of Study: 04/1987-1988; 1991-02/1992	Outcomes (ICD 9): All respiratory (460-519) Age groups analyzed: 0-14, 15-64, 65+, all ages Study Design: Timeseries N: 19,901 Statistical Analyses: APHEA protocol, Poisson regression GAM Covariates: Long-term trend, season, influenza, day of wk, holiday, temperature, humidity Season: Cool, Oct-Mar; Warm: Apr-Sep Dose-Response Investigated?: Yes Statistical Package: SAS Lag: 0,1,2 days, 0-3 cumulative avg.	NO_2 24-h avg: 37.3 ppb, Med: 35 SD = 13.8 IQR: 14 ppb 1-h max: 57.4 ppb, Med: 51 SD = 26.4 IQR: 21 ppb # of stations: 2	SO_2 r = 0.45 BS r = 0.44 O_3	Increment: 90th-10th percentile (24-h avg: 27 ppb) All year All ages 1.0114 [1.006, 1.0222] lag 2 0-14 yrs 1.0104 [0.9943, 1.0267] lag2 15-64 yr 1.0113 [0.9920, 1.0309] lag 1 ≥65 yr 1.0216 [1.0049, 1.0386] lag 2 Warm season All ages 1.0276 [1.0042, 1.0515] lag 2 0-14 yrs 1.038 [1.0009, 1.0765] lag 2 15-64 yr 1.0040 [0.9651, 1.0445] lag 1 >65 yr 1.0326 [0.9965, 1.0699] lag 2 Cool season All ages 1.0060 [0.9943, 10177] lag2 0-14 yrs 1.0027 [0.9855, 1.0202] lag2 15-64 yr 1.0136 [0.9920, 1.0357] lag 1 >65 yr 1.0174 [0.9994, 1.0358] lag 2

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Atkinson et al. (1999a) London, England Period of Study: 1992 to 1994 Days: 1096	Outcomes (ICD 9): All respiratory (460-519), Asthma (493), Asthma + COPD (490-6), Lower respiratory disease (466, 480-6) Age groups analyzed: 0-14, 15-64, 65+, all ages Study Design: Time series N: 165,032 Statistical Analyses: APHEA protocol, Poisson regression Covariates: Long-term trend, season, influenza, day of wk, holiday, temperature, humidity Season: Cool, Oct-Mar; Warm: Apr-Sep Dose-Response Investigated?: Yes Statistical Package: SAS Lag: 0,1,2 days, 0-1, 0-2, 0-3 cum. avg.	NO_2 1 h mean: 50.3 ppb, SD 17.0, Range: 22.0, 224.3 ppb, 10th centile: 34.3, 90th centile: 70.3 # of stations: 3, r = 0.7, 0.96	O_3, CO, PM_{10}, BS, SO_2	Increment: 36 ppb (90th-10th centile) All ages Respiratory 1.64% [0.14, 3.15] lag 1 Asthma 1.80% [−0.77, 4.44 lag 0 0-14 yrs Respiratory1.94% [−0.39, 4.32] lag 2 Asthma . 1% [−1.42, 5.77] lag 3 15-64 yrs Respiratory 1.61% [−0.82, 4.09] lag 1 Asthma 5.08% [0.81, 9.53] lag 1 65+ yrs Respiratory 2.53% [0.58, 4.52] lag 3 Asthma 4.53% [−2.36, 11.91] lag 3 COPD3.53% [0.64, 6.50] lag 3 Lower Resp. 3.47% [0.08, 6.97] lag 3

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Wong* et al. (2002) London England and Hong Kong Period of Study: London: 1992-1994 Hong Kong: 1995-1997 Days: 1,096	Outcomes (ICD 9): All respiratory admissions (460-519); asthma (493) Age groups analyzed: 15-64, 65+, all ages Study Design: Timeseries Statistical Analyses: APHEA protocol, Poisson regression with GAM Covariates: Long-term trend, season, influenza, day of wk, holiday, temperature, humidity, thunderstorms Season: Cool, Oct-Mar; Warm: Apr-Sep Dose-Response Investigated?: Yes Statistical Package: S-Plus Lag: 0,1,2,3,4 days, 0-1 cum. avg.	24 h NO_2 µg/m^3 Hong Kong Mean: 55.9 Warm: 48.1 Cool 63.8 SD 19.4 Range: 15.3, 151.5 10th: 31.8 50th: 53.5 90th: 81.8 London Mean: 64.3 Warm: 62.6 Cool 66.1 SD 20.4 Range: 23.7, 255.8 10th: 42.3 50th: 61.2 90th: 88.8 # of stations: Hong Kong: 7, r = 0.65, 0.90 London: 3, r = 0.80	Hong Kong PM_{10} r = 0.82 SO_2 r = 0.37 O_3 r = 0.43 London PM_{10} r = 0.68 SO_2 r = 0.71 O_3 r = −0.29	Increment: 10 µg/m^3 Asthma, 15-64 years Hong Kong ER −0.6 [−2.1, 1.0] lag 0-1 ER −1.3 [−2.6, 0.1] lag 1 Warm: ER −0.5 [−2.7, 1.6] lag 0-1 Cool: ER −0.6 [−2.8, 1.6] lag 0-1 London ER 1.0 [0.0, 2.1] lag 0-1 ER 1.1 [0.2, 2.0] lag 2 Warm: ER 0.6 [−0.8, 2.0] lag 0-1 Cool: ER 1.3 [−0.1, 2.8] lag 0-1 Respiratory 65+ years Hong Kong ER 1.8 [1.2, 2.4] lag 0-1 ER 1.3 [0.8, 1.8] lag 0 Warm: ER 0.8 [0.1, 1.6] lag 0-1 Cool: ER 3.0 [2.1, 3.9] lag 0-1 +O_3: ER 1.6 [1.0, 2.3] lag 0-1 +PM_{10}: ER 1.7 [0.8, 2.7] lag 0-1 +SO_2: ER 1.6 [0.8, 2.4] lag 0-1 London ER -0.1 [−0.6, 0.5] lag 0-1 ER 0.9 [0.5, 1.3] lag 3 Warm: ER 0.6 [−0.2, 1.4] lag 0-1 Cool: ER −0.7 [−1.4, 0.0] lag 0-1 +O_3: ER −0.1 [−0.5, 0.6] lag 0-1 +PM_{10}: ER −0.4 [−1.2, 0.4] lag 0-1 +SO_2: ER −0.2 [−0.9, 0.5] lag 0-1

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Anderson et al. (1998) London, England Period of Study: April 1987-February 1992 Days: 1,782	Outcomes (ICD 9): Asthma (493) Age groups analyzed: <15, 15-64, 65+ Study Design: Ti N: 16 Statistical Analyses: APHEA protocol, Poisson regression Covariates: Time trends, seasonal cycles, day of wk, public holidays, influenza epidemics, temperature, humidity Season: Cool (Oct-Mar); Warm (Apr-Sep) Dose-Response Investigated?: Yes Statistical Package: S Lag: 0,1,2 days	24-h avg NO_2 (ppb) Mean: 37.2 SD: 12.3 Range: 14, 182 5th: 22 10th: 25 25th: 30 50th: 36 75th: 42 90th: 50 95th: 58 1-h max NO_2 (ppb) Mean: 57.2 SD: 23.0 Range: 21, 370 5th: 35 10th: 38 25th: 44 50th: 52 75th: 64 90th: 81 95th: 98 Number of stations: 2	O_3 SO_2 BS	Increment: 10 ppb in 24 h NO_2 0-14 yrs Whole year RR 1.25 [0.3, 2.2] lag 2; RR 1.77 [0.39, 3.18] lag 0-3 + O_3 RR 1.13 [−0.10, 2.36] lag 2 + SO_2 RR 0.97 [−0.05, 1.99] lag 2 + BS RR 2.26 [0.83, 3.71] lag 2 Warm season RR 1.42 [−0.3, 3.17] lag 2; RR 3.01 [3.8, 5.72] lag 0-3 Cool season RR 1.18 [0.02, 2.35] lag 2; RR 1.22 [−0.48, 2.96] lag 0-3 15-64 yrs Whole year RR 0.95 [−0.26, 2.17] lag 0; RR 0.99 [−0.36, 3.36] lag 0-1 Warm RR 0.46 [−1.70, 2.67] lag 0; RR 0.05 [−2.45, 2.61] lag 0-1 Cool season RR 1.21 [−0.22, 2.5] lag 0; RR 1.43 [−0.18, 3.06] lag 0-1 65+ yrs Whole year RR 2.96 [0.67, 5.31] lag 2; RR 3.14 [−0.04, 6.42] lag 0-3 + O_3 RR 4.51 [1.43, 7.69] lag 2 + SO_2 RR 2.49 [−0.25, 5.31] lag 2 + BS RR 1.88 [−1.49, 5.36] lag 2 Warm RR 1.89 [−2.41, 6.38] lag 2; RR −1.76 [−7.27, 4.07] lag 0-3 Cool season RR 3.52 [0.81, 6.30] lag 2; RR 5.57 [1.85, 9.43] lag 0-3

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Anderson et al. (1998) (cont'd)				+ O_3 RR 5.14 [0.69, 9.79] lag 2 + SO_2 RR 2.10 [−1.08, 5.39] lag 2 + BS RR 4.47 [−0.04, 9.19] lag 2 All ages Whole year RR 1.25 [0.49, 2.02] lag 2; RR 2.05 [0.96, 3.15] lag 0-3 + O_3 RR 1.08 [0.12, 2.05] lag 2 + SO_2 RR 0.99 [0.18, 1.81] lag 2 + BS RR 1.23 [0.47, 2.00] lag 2 Warm RR 1.15 [−0.25, 2.57] lag 2; RR 1.54 [-0.54, 3.67] lag 0-3 Cool season RR 1.30 [0.38, 2.23] lag 2; RR 2.26 [0.94, 3.59] lag 0-3 + O_3 RR 0.50 [−0.79, 1.81] lag 2 + SO_2 RR 1.10 [0.12, 2.08] lag 2 + BS RR 1.29 [0.37, 2.22] lag 2
Prescott et al. (1998) Edinburgh, United Kingdom Period of Study: 10/92-6/95	Outcomes (ICD 9): Pneumonia (480-7), COPD + Asthma (490-496) Age groups analyzed: <65, 65+ Study Design: Time series Statistical Analyses: Poisson log linear regression Covariates: Trend, seasonal and wkly variation, temperature, wind speed, day of wk Lag: 0,1 or 3 day rolling avg	NO_2: 26.4 ± 7.0 ppb Min: 9 ppb Max: 58 ppb IQR: 10 ppb # of Stations: 1	CO PM_{10} SO_2 O_3 BS	Increment: 10 ppb Respiratory admissions >65 yrs 3.1 [−4.6, 11.5] rolling 3 day avg <65 yrs −0.2% [−7.5, 7.7] rolling 3 day avg

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Thompson et al. (2001) Belfast, Northern Ireland Period of Study: 1993-1995	Outcomes: Asthma ICD9: NR Age groups analyzed: 0-14 Study Design: Time series N: 1,095 Number of hospitals: 1 Statistical Analyses: Poisson regression Covariates: Season, long-term trend, temperature, day of wk, holidays Season: Warm (May-Oct), Cold (Nov-Apr) Statistical Package: Stata Lag: 0,1,2,3 days	24 h mean: Warm: 19.2 (7.9) ppb; range: 13-23 Cold: 23.3 (9.0) ppb; range: 18-28	SO_2 r = 0.82 PM_{10} r = 0.77 CO r = 0.69 O_3 r = –0.62 NOx r = 0.93 log (NO) r = 0.84 log (CO) r = 0.69	Increment: 10 ppb All seasons RR 1.08 [1.03, 1.13] lag 0 RR 1.11 [1.05, 1.17] lag 0-1 RR 1.10 [1.04, 1.17] lag 0-2 RR 1.12 [1.03, 1.02] lag 0-3 Warm season RR 1.14 [1.04, 1.26] lag 0-1 Cold season RR 1.10 [1.03, 1.17] lag 0-1 NO_2 + Benzene RR 0.99 [0.87, 1.13] lag 0-1 *Model made no allowance for possible autocorrelation in the data or for extra-Poisson variation

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Hagen et al. (2000) Drammen, Norway Period of Study: 1994-1997	Outcomes (ICD 9): All respiratory admissions (460-519) Age groups analyzed: All ages Study Design: Time series Number of hospitals: 1 Statistical Analyses: Poisson regression with GAM (adhered to HEI phase 1.B report) Covariates: Time trends, day of wk, holiday, influenza, temperature, humidity Lag: 0,1,2,3 days	NO_2 24-h avg ($\mu g/m^3$): 36.15, SD = 16 IQR: 16.92 $\mu g/m^3$ # of Stations: 2	PM_{10} r = 0.61 SO_2 r = 0.58 benzene r = 0.31 NO r = 0.70 O_3 r = −0.47 Formaldehyde r = 0.68 Toluene r = 0.65	Increment: NO_2: 16.92 $\mu g/m^3$ (IQR); NO: 29$\mu g/m^3$ (IQR) Single-pollutant model Respiratory disease only NO_2: RR 1.058 [0.994, 1.127] NO: 1.048 [1.013, 1.084] All disease NO_2: RR 1.011 [0.988, 1.035] Two-pollutant model with PM_{10} NO_2 1.044 [0.966, 1.127] NO: 1.045 [1.007, 1.084] Three-pollutant model with PM_{10} + Benzene NO_2 1.015 [0.939, 1.097] NO: 1.031 [0.986, 1.077]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Oftedal et al. (2003) Drammen, Norway Period of Study: 1994-2000	Outcomes (ICD 10): All respiratory admissions (J00-J99) Age groups analyzed: All ages Study Design: Time series Statistical Analyses: Semi-parametric Poisson regression, GAM with more stringent criteria Covariates: Temperature, humidity, influenza Lag: 2,3 days	Mean: 33.8 μg/m^3, SD = 16.2 IQR: 20.8 μg/m^3	PM_{10} SO_2 O_3 Benzene Formaldehyde Toluene	Increment: 20.8 μg/m^3 (IQR) All respiratory disease Single-pollutant model RR 1.060 [1.017, 1.105] lag 3 Two-pollutant model Adjusted for PM_{10} RR 1.063 [1.008, 1.120] Adjusted for benzene RR 1.046 [1.002, 1.091]
Pönkä and Virtanen (1994) Helsinki, Finland Period of Study: 1987-1989 Days: 1096	Outcomes (ICD 9): Chronic bronchitis and emphysema (493) Age groups analyzed: <65, ≥65 Study Design: Time series Statistical Analyses: Poisson regression Covariates: Season, day of wk, year, influenza, humidity, temperature Season: Summer (Jun-Aug), Autumn (Sep-Nov), Winter (Dec-Feb), Spring (Mar-May) Lag: 0-7 days	24 h mean: 39 μg/m^3, SD = 16.2; range: 4, 170 # of stations: 2	SO_2 O_3 TSP	Increment: NR Chronic bronchitis and emphysema >65 yrs RR 0.87 [0.71, 1.07] lag 0 RR 1.07 [0.86, 1.33] lag 1 RR 1.16 [0.93, 1.46] lag 2 RR 1.08 [0.86, 1.35] lag 3 RR 0.94 [0.76, 1.18] lag 4 RR 0.90 [0.72, 1.12] lag 5 RR 1.31 [1.03, 1.66] lag 6 RR 0.82 [0.67, 1.01] lag 7 <65 yrs NR

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Dab[+] et al. (1996) Paris, France Period of Study : 1/1/87-9/30/92	Outcomes (ICD 9): All respiratory (460-519), Asthma (493), COPD (490-496) Age groups analyzed: All ages Study Design: Time series Number of hospitals: 27 Statistical Analyses: Poisson regression, followed APHEA protocol Covariates: Temperature, relative humidity, influenza, long-term trend, season, holiday, medical worker strike Lag: 0,1,2 days, 0-3 cumulative	NO_2 24-h avg: 45 µg/m^3 5th: 22, 99th: 108.3 Daily maximum 1 h concentration: 73.8 µg/m^3 5th: 37.5, 99th: 202.7	SO_2 O_3 PM_{13} BS	Increment: 100 µg/m^3 All respiratory (1987-1990) 24-h avg NO_2 RR 1.043 [0.997, 1.090] lag 0 1-h max NO_2 RR 1.015 [0.993, 1.037] lag 0 Asthma (1987-1992) 24-h avg RR 1.175 [1.059, 1.304] lag 0-1 1-h max RR 1.081 [1.019, 1.148] lag 0-1 COPD 24-h avg RR 0.974 [0.898, 1.058] lag 2 1-h max RR 0.961 [0.919, 1.014] lag 2
Llorca et al. (2005) Torrelavega, Spain Period of Study: 1992-1995 Days: 1,461	Outcomes (ICD 9): All respiratory admissions (460-519) Age groups analyzed: All ages Study Design: Time series Number of hospitals: 1 Statistical Analyses: Poisson regression Covariates: Short and long-term trends Statistical Package: Stata Lag: NR	24-h avg NO_2: 21.3 µg/m^3, SD = 16.5 24-h avg NO: 12.2 µg/m^3, SD = 15.2 # of Stations: 3	SO_2 r = 0.588 NO r = 0.855 TSP r = -0.12 SH_2 r = 0.545	Increment: 100 µg/m^3 Single-pollutant model All cardio-respiratory admissions NO_2: RR 1.37 [1.26, 1.49] NO: RR 1.33 [1.22, 1.46] Respiratory admissions NO_2: RR 1.54 [1.34, 1.76] NO: RR 1.35 [1.17, 1.56] 5-pollutant model All cardio-respiratory admissions NO_2: RR 1.20 [1.05, 1.39] NO: RR 0.93 [0.79, 1.09] Respiratory admissions NO_2: RR 1.69 [1.34, 2.13] NO: RR 0.87 [0.67, 1.13]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Migliaretti and Cavallo (2004) Turin, Italy Period of Study: 1997-1999	Outcome(s) (ICD 9): Asthma (493) Age groups analyzed: <4, 4-15 Study Design: Case-Control Controls: age matched with other respiratory disease (ICD9: 460-7, 490-2, 494-6, 500-19) N: cases = 734, controls = 25,523 Statistical Analyses: logistic regression Covariates: seasonality, temperature, humidity, solar radiation Seasons: Cold: Oct-Mar; Warm: Apr-Sep Statistical Package: SPSS Lag: 0-3 days and cumulative	Controls: Mean: 113.3 μg/m^3, SD = 30.5 Cases: Mean: 117.4 μg/m^3, SD = 29.7	TSP	Increment: 10 μg/m^3 <4 yrs 2.8% [0.03, 5.03] lag 1-3 cumulative 4-15 yrs 2.7% [−0.01, 6.06] lag 1-3 cumulative All ages 2.8% [0.07, 4.09] lag 1-3 cumulative Two-pollutant model adjusted for TSP NO_2 2.1% [−0.1, 5.6]
Fusco* et al. (2001) Rome, Italy Period of Study : 1/1/95-10/31/97	Outcomes (ICD 9): All respiratory (460-519 excluding 470-478), Asthma (493), COPD (490-492, 494-496), Respiratory infections (460-466, 480-486) Age groups analyzed: 0-14, all ages Study Design: Time series Statistical Analyses: Semi-parametric Poisson regression with GAM Covariates: Influenza, day, temperature, humidity, day of wk, holiday Season: Warm (Apr-Sep), Cold (Oct – Mar) Statistical Package: S-Plus 4 Lag: 0-4 days	NO_2 24-h avg (μg/m^3): 86.7, SD = 16.2 IQR: 22.3 μg/m^3 # of stations: 5; r = 0.66-0.79	PM_{10}: All year r = 0.35 Cold r = 0.50 Warm r = 0.25 SO_2: All year r = 0.33 Cold r = 0.40 Warm r = 0.68 CO: All year r = 0.31 Cold r = 0.41 Warm r = 0.59 O_3: All year r = 0.19 Cold r = 0.19 Warm r = 0.13	Increment: 22.3 μg/m^3 (IQR) All respiratory All ages 2.5% [0.9, 4.2] lag 0 0-14 yrs 4.0% [0.6, 7.5] lag 0 Respiratory infections All ages 4.0% [1.6, 6.5] lag 0 0-14 yrs 4.0% [0.2, 8.0] lag 0 Asthma All ages 4.6% [−0.5, 10.0] lag 0 0-14 yrs 10.7% [3.0, 19.0] lag 1 COPD ≥65 yrs 2.2% [−0.7, 5.2] lag 0 Multipollutant models All respiratory (NO_2 + CO)

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
EUROPE (cont'd)				
Fusco* et al. (2001) (cont'd)				All ages : 0.9% [−0.8, 2.8] lag 0 0-14 yrs : 3.3% [−0.2, 6.9] lag 0 Acute infections (NO_2 + CO) All ages : 3.9% [1.3, 6.7] lag 0 0-14 yrs : 2.9% [−1.0, 7.0] lag 0 Asthma (NO_2 + CO) All ages : 1.4% [−3.9, 7.1] lag 0 0-14 yrs : 8.3% [−0.1, 17.4] lag 1 COPD (NO_2 + CO) ≥65 yrs: −1.0%[−4.1, 2.2] lag 0
Pantazopoulou et al. (1995) Athens, Greece Period of Study : 1988	Outcomes: All respiratory admissions ICD9: NR Age groups analyzed: All ages Study Design: Time series N: 15,236 Number of hospitals: 14 Statistical Analyses: Multiple linear regression Covariates: Season, day of wk, holiday, temperature, relative humidity Season: Warm (3/22-9/21), Cold (1/1-3/21 and 9/22-12/31) Lag: NR	NO_2 24-h avg Winter: 94 μg/m^3, SD = 25 5th: 59, 50th: 93, 95th: 135 Summer: 111 μg/m^3, SD = 32 5th: 65, 50th: 108, 95th: 173 # of stations: 2	CO BS	Increment: 76 μg/m^3 in winter and 108 μg/m^3 in summer (95th-5th) Respiratory disease admissions Winter: Percent increase: 24% [6.4, 43.5] Summer: Percent increase: 9.3% [−14.1, 24.4]

TABLE AX6.3.1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
LATIN AMERICA				
Gouveia and Fletcher, (2000a) São Paulo, Brazil Period of Study: 11/92-9/94	Outcomes (ICD 9): All respiratory; Pneumonia (480-486); asthma or bronchitis (466, 490, 491, 493) Age groups analyzed: <1; <5 years Study Design: Time series Statistical Analyses: Poisson regression Covariates: Long-term trend, season, temperature, relative humidity, day of wk, holiday, strikes in public transport or health services Season: Cool (May-Oct), Warm (Nov-Apr) Statistical Package: SAS Lag: 0, 1, 2 days	1-h max NO_2 ($\mu g/m^3$) Mean: 174.3 SD: 101.3 Range: 26.0, 692.9 5th: 62.0 25th: 108.8 50th: 151.7 75th: 210.0 95th: 388.0 # of stations: 4	SO_2 r = 0.37 PM_{10} r = 0.40 CO r = 0.35 O_3 r = 0.25	Increment: 319.4 $\mu g/m^3$ (90th-10th) All Respiratory <5 years RR 1.063 [0.999, 1.132] lag 0 <5 years + O_3 RR 1.050 [0.985, 1.120] <5 years + PM_{10} RR 1.043 [0.972, 1.119] <5 years + O_3+ PM_{10} RR 1.035 [0.963, 1.113] <5 years Cool RR 1.04 [0.96, 1.11] (estimated from graph) <5 years Warm RR 1.09 [1.01, 1.16] (estimated from graph) Pneumonia <5 years RR 1.093 [1.016, 1.177] lag 0 <1 year RR 1.091 [0.996, 1.193] lag 0 Asthma <5 years RR 1.107 [.0.940, 1.300] lag 2
Braga* et al. (2001) São Paulo, Brazil Period of Study: 1/93-11/97	Outcomes (ICD 9): All respiratory admissions (460-519) Age groups analyzed: 0-19, ≤2, 3-5, 6-13, 14-19 Study Design: Time series Statistical Analyses: Poisson regression with GAM Covariates: Long-term trend, season, temperature, relative humidity, day of wk, holiday Statistical Package: S-Plus 4.5 Lag: 0-6 moving avg	NO_2 mean: 141.4 $\mu g/m^3$, SD = 71.2 IQR: 80.5 $\mu g/m^3$ Range: 25, 652.1 # of stations: 5-6	PM_{10} r = 0.62 SO_2 r = 0.54 CO r = 0.58 O_3 r = 0.34	Increment: 80.5 $\mu g/m^3$ (IQR) All Respiratory admissions <2 yrs 9.4% [6.2, 12.6] lag 5 3-5 yrs 1.6% [−6.4, 9.6] 6-13 yrs 2.3% [−5.9, 10.4] 14-19 yrs -3.0% [−15.7, 9.7] All ages 6.5% [3.3, 9.7]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
LATIN AMERICA (cont'd)				
Farhat* et al. (2005) São Paulo, Brazil Period of Study: 8/96-8/97 Days: 396	Outcomes (ICD 9): Pneumonia/bronchiopheumonia (480-6), asthma (493), bronchiolitis (466), Obstructive disease 493, 466) Age groups analyzed: <13 Study Design: Time series N: 1,021 Number of hospitals: 1 Statistical Analyses: Poisson regression with GAM Covariates: Time, temperature, humidity, day of wk, season Statistical package: S-Plus Lag: 0-7 days, 2,3,4 day moving avg	Mean: 125.3 μg/m^3 SD = 51.7 IQR: 65.04 μg/m^3 Range: 42.5, 369.5	PM_{10} r = 0.83 SO_2 r = 0.66 CO r = 0.59 O_3 r = 0.47	Increment: 65.04 μg/m^3 (IQR) Single pollutant models (estimated from graphs) Asthma: ≈ 32% [8,56] lag 0-2 Pneumonia: ≈ 17.5% [2.5, 32.5] lag 0-3 Asthma or Bronchiolitis NO_2 + PM_{10} 47.7% [1.15, 94.2] lag 0-2 NO_2 + SO_2 33.1% [5.7, 60.5] lag 0-2 NO_2 + CO 28.8% [−0.2, 57.9] lag 0-2 NO_2 + O_3 28.0% [−1.0, 57.0] lag 0-2 Multipollutant model (PM_{10}, SO_2, CO, O_3) 39.3% [−14.9, 93.5] 2 day avg. Pneumonia or bronchopneumonia NO_2 + PM_{10} 8.11% [−11.4, 27.6] lag 0-2 NO_2 + SO_2 13.1% [−3.4, 29.7] lag 0-2 NO_2 + CO 12.4% [−5.6, 30.4] lag 0-2 NO_2 + O_3 14.6% [−4.9, 34.1] lag 0-2 Multipollutant model (PM_{10}, SO_2, CO, O_3) 1.8% [−23.9, 27.6] lag 0-2

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
ASIA				
Lee et al. (2006) Hong Kong, China Period of Study: 1997-2002 Days: 2,191	Outcomes (ICD 9): Asthma (493) Age groups analyzed: ≤18 Study Design: Time series N: 26,663 Statistical Analyses: Semi-parametric Poisson regression with GAM (similar to APHEA 2) Covariates: Long-term trend, temperature, relative humidity, influenza, day of wk, holiday Statistical package: SAS 8.02 Lag: 0-5 days	NO_2 24 h mean: 64.7 µg/m^3, SD = 20.9 IQR: 27.1 µg/m^3 25th: 49.7, 75th: 76.8 # of stations: 9-10, r = 0.53, 0.94, mean = 0.78	PM_{10} r = 0.78 $PM_{2.5}$ r = 0.75 SO_2 r = 0.49 O_3 r = 0.35	Increment: 27.1 µg/m^3 (IQR) Asthma Single-pollutant model 4.37% [2.51, 6.27] lag 0 5.88% [4.00, 7.70] lag 1 7.19% [5.37, 9.04] lag 2 9.08% [7.26, 10.93] lag 3 7.64% [5.84, 9.48] lag 4 6.40% [4.60, 8.22] lag 5 Multipollutant model – including PM, SO_2, and O_3 5.64% [3.21, 8.14] lag 3 Other lags NR
Chew et al. (1999) Singapore Period of Study: 1990-1994	Outcome(s) (ICD 9): Asthma (493) Age groups analyzed: 3-12, 13-21 Study Design: Time series N: 23,000 # of Hospitals: 2 Statistical Analyses: Linear regression, GLM Covariates: variables that were significantly associated with ER visits were retained in the model Statistical Package: SAS/STAT, SAS/ETS 6.08 Lag: 1,2 days avgs	24-h avg: 18.9 µg/m^3, SD = 15.0, max < 40 # of Stations: 15	SO_2; r = −0.22 O_3; r = 0.17 TSP; r = 0.23	Categorical analysis (via ANOVA) p-value and Pearson correlation coefficient (r) using continuous data comparing daily air pollutant levels and daily number of hospital admissions Age Group: 3-12 13-21 Lag 0 r = 0.13 r = 0.05 p = 0.013 p < 0.18 Lag 1 r = 0.13 r = 0.02 P = 0.02 p = 0.75 Lag 2 r = 0.13 r = 0.07 p = 0.012 p = 0.35

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
ASIA (cont'd)				
Tsai et al. (2006) Kaohsiung, Taiwan Period of Study: 1996-2003 Days: 2922	Outcomes (ICD 9): Asthma (493) Study Design: Case-crossover N: 17,682 Statistical Analyses: Conditional logistic regression Covariates: Temperature, humidity Season: Warm (≥25ºC); Cool (<25ºC) Statistical package: SAS Lag: 0-2 days cumulative	NO_2 24 h mean: 27.20 ppb IQR: 17 ppb Range: 4.83, 63.40 # of stations: 6	PM_{10} SO_2 O_3 CO	Increment: 17 ppb (IQR) Seasonality Single-pollutant model >25°C 1.259 [1.111, 1.427] lag 0-2 <25°C 2.119 [1.875, 2.394] lag 0-2 Dual-pollutant model Adjusted for PM_{10} >25°C 1.082 [0.913, 1.283] lag 0-2 <25°C 2.105 [1.791, 2.474] lag 0-2 Adjusted for CO >25°C 0.949 [0.792, 1.137] lag 0-2 <25°C 2.30 [1.915, 2.762] lag 0-2 Adjusted for SO_2 >25°C 1.294 [1.128, 1.485] lag 0-2 <25°C 2.627 [2.256, 3.058] lag 0-2 Adjusted for O_3 >25°C 1.081 [0.945, 1.238] lag 0-2 <25°C 2.096 [1.851, 2.373] lag 0-2
Lee* et al. (2002) Seoul, Korea Period of Study: 12/1/97-12/31/99 Days: 822	Outcomes (ICD 10): Asthma (J45 – J46) Age groups analyzed: <15 Study Design: Time series N: 6,436 Statistical Analyses: Poisson regression, log link with GAM Covariates: Time, day of wk, temperature, humidity Season: Spring (Mar-May), Summer (Jun-Aug), Fall (Sep-Nov), Winter (Dec-Feb) Statistical package: NR Lag: 0-2 days cumulative	24 h NO_2 (ppb) Mean: 31.5 SD: 10.3 5th: 16.0 25th: 23.7 50th: 30.7 75th: 38.3 95th: 48.6 # of stations: 27	SO_2 r = 0.72 O_3 r = −0.07 CO r = 0.79 PM_{10} r = 0.74	Increment: 14.6 ppb (IQR) Asthma NO_2 RR 1.15 [1.10, 1.20] lag 0-2 $NO_2 + PM_{10}$ RR 1.13 [1.07, 1.19] lag 0-2 $NO_2 + SO_2$ RR 1.20 [1.11, 1.29] lag 0-2 $NO_2 + O_3$ RR 1.14 [1.09, 1.20] lag 0-2 NO_2 + CO RR 1.12 [1.03, 1.22] lag 0-2 $NO_2 + O_3 + CO + PM_{10} + SO_2$ RR 1.098 [1.002, 1.202]

TABLE AX6.3-1 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: HOSPITAL ADMISSIONS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants & Correlations	Effects: Relative Risk or Percent Change & Confidence Intervals (95%)
ASIA (cont'd)				
Wong et al. (1999) Hong Kong, China Period of Study: 1994-1995	Outcomes (ICD 9): All respiratory admissions (460-6, 471-8, 480-7, 490-6); Asthma (493), COPD (490-496), Pneumonia (480-7) Age groups analyzed: 0-4, 5-64, ≥65, all ages # of hospitals: 12 Study Design: Time series Statistical Analyses: Poisson regression (followed APHEA protocol) Covariates: Trend, season, day of wk, holiday, temperature, humidity Statistical package: SAS 8.02 Lag: days 0-3 cumulative	Median 24 h NO_2: 51.39 µg/m^3 Range: 16.41, 122.44 25th: 39.93, 75th: 66.50 # of stations: 7, r = 0.68, 0.89	O_3 SO_2 PM_{10} r = 0.79	Increment = 10 µg/m^3 Overall increase in admissions: 1.020 [1.013, 1.028] lag 0-3 Respiratory Relative Risks (RR) 0-4 yrs: 1.020 [1.010, 1.030] lag 0-3 5-64yrs: 1.023 [1.011, 1.034] lag 0-3 >65 yrs: 1.024 [1.014, 1.035] lag 0-3 Cold Season: 1.004 [0.988, 1.020] NO_2 + high PM_{10}: 1.009 [0.993, 1.025] NO_2 + high O_3: 1.013 [0.999, 1.026] Asthma: 1.026 [1.01, 1.042] lag 0-3 COPD: 1.029 [1.019, 1.040] lag 0-3 Pneumonia: 1.028 [1.015, 1.041] lag 0-3
Wong et al. (2001a) Hong Kong, China Period of Study: 1993-1994	Outcomes (ICD 9): Asthma (493) Age groups analyzed: ≤15 N: 1,217 # of hospitals: 1 Study Design: Time series Statistical Analyses: Poisson regression (followed APHEA protocol) Covariates: Season, temperature, humidity Season: Summer (Jun-Aug), Autumn (Sep-Nov), Winter (Dec-Feb), Spring (Mar-May) Lag: 0,1,2,3,4,5 days; and cumulative 0-2 and 0-3 days.	24-h avg NO_2 mean: 43.3 µg/m^3, SD = 16.6 Range: 9, 106 µg/m^3 Autumn: 51.7 (17.6) Winter: 46.6 (15.5) Spring: 40.7 (11.8) Summer: 32.6 (13.7) # of stations: 9	PM_{10} SO_2	Increment: 10 µg/m^3 Asthma All year: 1.08 p = 0.001 Autumn: 1.08 p = 0.017 Winter: NR Spring: NR Summer: NR

*Default GAM

+Did not report correction for over-dispersion

NR: Not Reported

APHEA: Air Pollution and Health: A European Approach

TABLE AX6.3-2. RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
UNITED STATES				
Jaffe et al. (2003) 3 cities, Ohio, United States (Cleveland, Columbus, Cincinnati) Period of Study: 7/91-6/96	Outcome (ICD-9): Asthma (493) Age Groups Analyzed: 5-34 Study Design: Time series N: 4,416 Statistical Analyses: Poisson regression using a standard GAM approach Covariates: city, day of wk, wk, yr, minimum temperature, overall trend, dispersion parameter Season: June to August only Dose-response investigated: Yes Statistical Package: NR Lag: 0-3 days	Cincinnati 24-h avg: 50 ppb, SD = 15 Cleveland 24-h avg: 48 ppb, SD = 16 NO_2 was not monitored in Columbus due to relatively low levels	Cincinnati: PM_{10}; r = 0.36 SO_2; r = 0.07 O_3; r = 0.60 Cleveland: PM_{10}; 0.34 SO_2; r = 0.28 O_3; r = 0.42 No multipollutant models were utilized.	Increment: 10 ppb Cincinnati: 6% [−1.0, 13] lag 1 Cleveland: 4% [−1, 8] lag 1 All cities: 3% [−1.0, 7] Attributable risk from NO_2 increment: Cincinnati 0.72 (RR 1.06) Cleveland 0.44 (RR 1.04) Regression diagnostics for Cincinnati showed significant linear trend during entire study period and for each wk (6/1-8/31). No trends observed for Cleveland. Regression Models assessing exposure thresholds showed a possible dose-response for NO_2 (percent increase after 40 ppb). No increased risk until minimum concentration of 40 ppb was reached.
Norris* et al. (1999) Seattle, WA, United States Period of Study: 1995-1996	Outcome (ICD-9): Asthma (493) Age groups analyzed: <18 yrs Study Design: Time series N: 900 ER visits Statistical Analyses: Semi-parametric Poisson regression using GAM. Covariates: day of wk, time trends, temperature, dew point temperature Dose-response investigated: Yes Statistical Package: NR Lag: 0,2 days	24 h: 20.2 ppb, SD = 7.1 IQR: 9 ppb 1-h max: 34.0 ppb, SD = 11.3 IQR: 12 ppb	CO; r = 0.66 PM; r = 0.66 SO_2; r = 0.25	Increment: IQR 24-h avg (9 ppb increment) RR 0.99 [0.90, 1.08] lag 2 1-h max (12 ppb increment) RR 1.05 [0.99, 1.12] lag 0 Age and hospital utilization (high and low) segregation (<5, 5-11, and 12-17 yrs) did not figure significantly in the association between emergency room visits and asthma.

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Lipsett et al. (1997) Santa Clara County, California, United States Period of Study: 1988-1992	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: All Study Design: Time series Statistical Analyses: Poisson Regression; GEE repeated with GAM Covariates: minimum temperature, day of study, precipitation, hospital, day of wk, yr, overdispersion parameter Season: Winters only Statistical Package: SAS, S-Plus, Stata Lag: 0-5 days	NO_2 1-h mean: 69 ppb, SD = 28 Range: 29, 150 ppb	PM_{10}; r = 0.82 COH; r = 0.8 No multipollutant model due to high correlation between pollutants	Same day NO_2 was associated with ER visits for asthma ($\beta = 0.013$, $p = 0.024$) Absence of association between lagged or multiday specifications of NO_2 and asthma ER visits (data not shown) suggest that same day association may be artifact of covariation with PM_{10}

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Peel et al. (2005) Atlanta, GA, United States Period of Study: 1/93-8/2000	Outcome(s) (ICD-9): All respiratory (460-6, 477, 480-6, 480-6, 490-3, 496); Asthma (493); COPD (491-2, 496); Pneumonia (480-486); Upper Respiratory Infection (460-6, 477) Age groups analyzed: All, 2-18 Study Design: Time series N: 484,830 # of Hospitals: 31 Statistical Analyses: Poisson Regression, GEE, GLM, and GAM (data not shown for GAM) Covariates: day of wk, hospital entry/exit, holidays, time trend; season, temperature, dew point temperature Statistical Package: SAS, S-Plus Lag: 0 to 7 days. 3 day moving avgs.	1-h max: 45.9 ppb, SD = 17.3	O_3; r = 0.42 SO_2; r = 0.34 CO; r = 0.68 PM_{10}; r = 0.46 Evaluated multipollutant models (data not shown)	Increment: 20 ppb All respiratory RR 1.016 [1.006, 1.027] lag 0-2, 3 day moving avg Upper Respiratory Infection (URI) RR 1.019 [1.006, 1.031] lag 0-2, 3 day moving avg Asthma All: 1.014 [0.997, 1.030] lag 0-2, 3 day moving avg 2-18: 1.027 [1.005, 1.050] lag 0-2, 3 day moving avg Pneumonia RR 1.000 [0.983, 1.019] lag 0-2, 3 day moving avg COPD RR 1.035 [1.006, 1.065] lag 0-2, 3 day moving avg

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Tolbert et al. (2000) Atlanta, GA, United States Period of Study: 1993-1995	Outcome(s) (ICD-9): Asthma (493), wheezing (786.09), Reactive airways disease (RADS) (519.1) Age groups analyzed: 0-16; 2-5, 6-10, 11-16 Study Design: Case-Control N: 5,934 Statistical Analyses: Ecological GEE analysis (Poisson model with logit link) and logistic regression Covariates: day of wk, day of summer, yr, interaction of day of summer and yr Season: Summers only Statistical Package: SAS Lag: 1 day (a priori)	NO_x 1-h max continuous Mean: 81.7 ppb, SD = 53.8 Range = 5.35, 306 Number of stations: 2	PM_{10}; r = 0.44 O_3; r = 0.51	Increment: 50 ppb Age 0-16: RR 1.012 [0.987, 1.039] lag 1

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
UNITED STATES (cont'd)				
Cassino* et al. (1999) New York City, NY United States Period of Study: 1/1989-12/1993	Outcome(s) (ICD-9): Asthma (493); COPD (496), bronchitis (490), emphysema (492), bronchiectasis (494) Study Design: Time series N: 1,115 # of Hospitals: 11 Statistical Analyses: Time series regression, Poisson regression with GLM and GAM; Linear regression, Logistic regression with GEE Covariates: Season, trend, day of wk, temperature, humidity Statistical Package: S Plus and SAS Lag: 0-3 days	24-h avg NO_2: Mean: 45.0 ppb Median: 43 ppb 10% 31 ppb 25% 37 ppb 75% 53 ppb 90% 63 ppb	O_3 CO SO_2	Increment: 15 ppb (IQR) RR 0.97 [0.85, 1.09] lag 0 RR 1.04 [0.92, 1.18] lag 1 RR 1.06 [0.94, 1.2] lag 2 RR 0.97 [0.86, 1.09] lag 3

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
CANADA				
Stieb et al. (1996) St. John, New Brunswick, Canada Period of Study: 1984-1992 (May-Sept only)	Outcome(s): Asthma ICD-9 Codes: NR Age groups analyzed: 0-15, >15 Study Design: Time series N: 1,163 # of Hospitals: 2 Statistical Analyses: SAS NLIN (Equivalent to Poisson GEE) Covariates: day of wk, long-term trends, Season: Summers only (May-Sep) Dose-response investigated?: Yes Statistical Package: SAS Lag: 0-3 days	1-h max NO_2 (ppb) Mean: 25.2 Range: 0, 120 95th 60	O_3 r = 0.16 SO_2 r = −0.03 SO_4^{2-} r = 0.16 TSP r = 0.15	Increment: NR $NO_2 + O_3$: β = −0.0037 (0.0023) lag 2
Stieb* et al. (2000) Saint John, New Brunswick, Canada Period of Study: Retrospective: 7/92-6/94 Prospective: 7/94-3/96	Outcome(s): Asthma; COPD; Respiratory infection (bronchitis, bronchiolitis, croup, pneumonia); All respiratory ICD-9 Codes: NR Age groups analyzed: All Study Design: Time series N: 19,821 Statistical Analyses: Poisson regression, GAM Covariates: day of wk, selected weather variables in each model Seasons: All yr, summer only Dose-Response investigated: Yes Statistical Package: S-Plus Lag: all yr = 0; summer only = 8	Annual mean: 8.9 ppb spring/fall mean: 10.0 ppb Max: 82	O_3; r = -0.02 SO_2; r = 0.41 TRS; r = 0.16 PM_{10}; r = 0.35 $PM_{2.5}$; r = 0.35 H^+; r = 0.25 SO_4^{2-}; r = 0.33 COH; r = 0.49 Assessed multipollutant models	Increment: 8.9 ppb (IQR) Respiratory visits: −3.8%, p = 0.070 lag 0 May to Sept: 11.5%, p = 0.17 lag 8 Multipollutant model (NO_2, O_3, SO_2) −3.6% [−7.5, 0.5] lag 0 Multipollutant model (ln(NO_2), O_3, SO_2 COH) May to Sept: 4.7% [0.8 to 8.6] lag 8 Non-linear effect of NO_2 on summertime respiratory visits observed and log transformation strengthened the association.

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST				
Sunyer et al. (1997) Multi-city, Europe (Barcelona, Helsinki, Paris, London) Period of Study: 1986-1992	Outcomes (ICD-9): Asthma (493) Age groups analyzed: <15, 15-64 Study Design: Time series Statistical Analyses: APHEA protocol, Poisson regression, GEE; meta-analysis Covariates: Humidity, temperature, influenza, soybean, long-term trend, season, day of wk Season: Cool, Oct-Mar; Warm: Apr-Sep Statistical Package: NR Lag: 0,1,2,3 and cumulative 1-3	24-h median (range) ($\mu g/m^3$) Barcelona: 53 (5, 142) Helsinki: 35 (9, 78) London: 69 (27, 347) Paris: 42 (12, 157) # of stations: Barcelona: 3 London: 2 Paris: 4 Helsinki: 8	SO_2 black smoke O_3	Increment: 50 $\mu g/m^3$ of 24-h avg for all cities combined Asthma 15-64 yrs 1.029 [1.003, 1.055] lag 0-1 1.038 [1.008, 1.068] lag 0-3, cumulative <15 yrs 1.026 [1.006, 1.049] lag 2 1.037 [1.004, 1.067] lag 0-3, cumulative 1.080 [1.025, 1.140] - Winter only Two-pollutant models: NO_2/Black smoke 15-64 yrs 1.055 [1.005, 1.109] lag 0-1 15-64 yrs 1.088 [1.025, 1.155] cumulative 0-3 <15 yrs 1.036 [0.956, 1.122] NO_2/SO_2 <15 yrs 1.034 [0.988, 1.082]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Atkinson et al. (1999b) London, United Kingdom Period of Study: 1/92-1294	Outcome(s) (ICD-9): Respiratory ailments (490-496), including asthma, wheezing, inhaler request, chest infection, COPD, difficulty in breathing, cough, croup, pleurisy, noisy breathing Age groups analyzed: 0-14; 15-64; ≥65; All ages Study Design: Time series N: 98,685 # of Hospitals: 12 Statistical Analyses: Poisson regression, APHEA protocol Covariates: long-term trend, season, day of wk, influenza, temperature, humidity Statistical Package: SAS Lag: 0,1,0-2, and 0-3 days	1-h max: 50.3 ppb, SD = 17.0 # of Stations: 3; r = 0.70, 0.96	NO_2, O_3 (8 h), SO_2 (24 h), CO (24 h), PM_{10} (24 h), BS	Increment: 36 ppb in 1-h max Single-pollutant model Asthma Only 0-14 yrs 8.97% [4.39, 13.74] lag 1 15-64 yrs 4.44% [0.14, 8.92] lag 1 All ages 4.37% [1.32, 7.52] lag 0 All Respiratory 0-14 yrs 2.17% [−0.49, 4.91] lag 1 15-64 yrs 1.87% [−0.69, 4.49] lag 2 >65 yrs 3.97% [0.51, 7.55] lag 0 All Ages 1.20% [−0.57, 3.00] Two-pollutant model Asthma Only 0-14 yrs: SO_2: 5.75% [0.39, 11.40] lag 1 CO: 8.34% [3.61, 13.29] lag 0 PM_{10}: 6.95% [1.96, 12.19] lag 2 BS: 8.32% [3.56, 13.30] lag 2 O_3: 9.68% [5.02, 14.54] lag 0

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Buchdahl et al. (1996) London, United Kingdom Period of Study: 3/1/92-2/28/93	Outcomes: Daily acute wheezy episodes ICD-9: NR Age groups analyzed: ≤16 Study Design: Case-control N: 1,025 cases, 4,285 controls Number of hospitals: 1 Statistical Analyses: Poisson regression Covariates: Season, temperature, wind speed Season: Spring (Apr-Jun), Summer (Jul-Sep), Autumn (Oct-Dec), Winter (Jan-Mar) Statistical Package: Stata Lag: 0-7 days	NO_2 24-h yr round mean: 60 μg/m^3, SD = 17 IQR: 17 μg/m^3 Spring: 59 (19) Summer: 55 (18) Fall: 66 (13) Winter: 61 (17)	SO_2 r = 0.62 O_3 r = −0.18	Increment: 17 μg/m^3 (IQR) No adjustments to model RR 1.07 [1.01, 1.14] lag not specified Adjusted for temperature and season. RR 1.02 [0.96, 1.09] lag not specified

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Thompson et al. (2001) Belfast, Northern Ireland Period of Study: 1993-1995	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: Children Study Design: Time series N: 1,044 Statistical Analyses: Followed APHEA protocol, Poisson regression analysis Covariates: Season, long-term trend, temperature, day of wk, holiday Season: Warm (May-Oct); Cold (Nov-Apr) Statistical Package: Stata Lag: 0-3	Warm Season NO_2 (ppb): Mean: 19.20; SD: 7.90; IQR: 13.0, 23.0 NO_x (ppb): Mean: 35.50; SD: 25.50; IQR: 21.0, 40.0 NO (ppb): Mean: 16.4; SD: 19.70; IQR: 7.0, 17.0 Cold Season NO_2 (ppb): Mean: 23.30; SD: 9.00; IQR: 18.0, 28.0 NO_x (ppb): Mean: 50.50; SD: 50.50; IQR: 26.0, 56.0 NO (ppb): Mean: 27.30; SD: 43.10; IQR: 9.0, 28.0	NO_2: PM_{10} r = 0.77 SO_2 r = 0.82 NO_x r = 0.93 NO r = 0.84 O_3 r = −0.62 CO r = 0.69 Benzene r = 0.83 NO_x: PM_{10} r = 0.73 SO_2 r = 0.83 NO_2 r = 0.92 NO r = 0.97 O_3 r = −0.73 CO r = 0.74 Benzene r = 0.86 NO: PM_{10} r = 0.65 SO_2 r = 0.76 NO_x r = 0.97 NO_2 r = 0.84 O_3 r = −0.76 CO r = 0.71 Benzene r = 0.82	NO_2 Increment: 10 ppb NO_x Increment: per doubling NO Increment: per doubling NO_2 Lag 0 RR 1.08 [1.03, 1.13] Lag 0-1 RR 1.11 [1.05, 1.17] Lag 0-2 RR 1.10 [1.04, 1.17] Lag 0-3 RR 1.12 [1.03, 1.20] Warm only Lag 0-1 RR 1.14 [1.04, 1.26] Cold only Lag 0-1 RR1.10 [1.03, 1.17] Adjusted for Benzene Lag 0-1 RR 0.99 [0.87, 1.13] NO_x Lag 0 RR 1.07 [1.02, 1.12] Lag 0-1 RR 1.10 [1.05, 1.16] Lag 0-2 RR 1.10 [1.03, 1.17] Lag 0-3 RR 1.11 [1.04, 1.20] Warm only Lag 0-1 RR 1.13 [1.03, 1.24] Cold only Lag 0-1 RR 1.09 [1.02, 1.16] Adjusted for Benzene Lag 0-1 RR 0.89 [0.77, 1.03] NO Lag 0 RR 1.04 [1.01, 1.07] Lag 0-1 RR 1.07 [1.03, 1.11] Lag 0-2 RR 1.06 [1.02, 1.11] Lag 0-3 RR 1.08 [1.02, 1.14] Warm only Lag 0-1 RR 1.08 [1.01, 1.16] Cold only Lag 0-1 RR 1.06 [1.01, 1.11] Adjusted for Benzene Lag 0-1 RR 0.93 [0.85, 1.01]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Boutin-Forzano et al. (2004) Marseille, France Period of Study: 4/97-3/98	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: 3-49 Study Design: Case-Crossover N: 549 Statistical Analyses: Logistic regression Covariates: minimal daily temperature, maximum daily temperature, minimum daily relative humidity, maximum daily relative humidity, day of wk Statistical Package: NR Lag: 0-4 days	Mean NO_2: 34.9 µg/m^3 Range: 3.0, 85	SO_2; r = 0.56 O_3; r = 0.58	Increment: 10 µg/m^3 Increased ER visits OR 1.0067 [0.9960, 1.0176] lag 0

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Castellsague et al. (1995) Barcelona, Spain Period of Study: 1986-1989	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: 15-64 Study Design: Time series # of Hospitals: 4 Statistical Analyses: Poisson regression Covariates: long-time trend, day of wk, temperature, relative humidity, dew point temperature Seasons: Winter : Jan-Mar; Summer: Jul-Sep Dose-Response investigated: Yes Statistical Package: NR Lag: 0, 1-5 days and cumulative. Summer: lag 2 days Winter: lag 1 day	Mean NO_2 ($\mu g/m^3$) Summer: 104.0 Winter: 100.8 IQR ($\mu g/m^3$): Summer: 48 Winter: 37 # of Stations: 15 manual, 3 automatic	SO_2; r = NR O_3; r = NR	Increment: 25 $\mu g/m^3$ Seasonal differences Summer: 1.071 [1.101, 1.130] lag 0-5 cumulative 1.045 [1.009, 1.081] lag 0 Winter: 1.072 [1.010, 1.137] lag 0-2 cumulative 1.056 [1.011, 1.104] lag 0 Asthma visits increased across quartiles of NO_2 in summer; a positive but less consistent increase across quartiles was observed in winter.

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Tobías et al. (1999) Barcelona, Spain Period of Study: 1986-1989	Outcome(s): Asthma ICD-9: NR Age groups analyzed: >14 Study Design: Time series Statistical Analyses: Poisson regression, followed APHEA protocol Covariates: temperature, humidity, long-term trend, season, day of wk Statistical Package: NR Lag: NR	24-h-avg NO_2 µg/m^3 Non-epidemic days: 54.7 (20.8) Epidemic days: 58.9 (26.7)	BS SO_2 O_3	$\beta \times 10^4$ (SE $\times 10^4$) using Std Poisson Without modeling asthma epidemics: 11.25 (11.79) $p > 0.1$ Modeling epidemics with 1 dummy variable: 1.18 (7.59) $p > 0.1$ Modeling epidemics with 6 dummy variables: 13.60 (7.79) $p < 0.1$ Modeling each epidemic with dummy variable: 14.40 (7.44) $p < 0.1$ $\beta \times 10^4$ (SE $\times 10^4$) using Autoregressive Poisson Without modeling asthma epidemics: 13.65 (11.81) $p > 0.1$ Modeling epidemics with 1 dummy variable: 3.28 (7.77) $p > 0.1$ Modeling epidemics with 6 dummy variables: 16.49 (8.01) $p < 0.05$ Modeling each epidemic with dummy variable: 18.18 (8.01) $p < 0.1$

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Galán et al. (2003) Madrid, Spain Period of Study: 1995-1998	Outcome(s) (ICD-9): Asthma (493) Age groups analyzed: All Study Design: Time series N: 4,827 Statistical Analyses: Poisson regression, (1) classic APHEA protocol and (2) GAM with stringent criteria Covariates: trend, yr, season, day of wk, holidays, temperature, humidity, influenza, acute respiratory infections, pollen Statistical Package: NR Lag: 0-4 days	24-h mean: 67.1 μg/m^3 SD = 18.0 IQR: 20.5 Max: 147.5 # of Stations: 15	PM_{10}; r = 0.717 SO_2; r = 0.610 O_3; r = −0.209	Increment: 10 μg/m^3 Asthma: RR 1.013 [0.991, 1.035] lag 0 RR 1.011 [0.989, 1.032] lag 1 RR 1.013 [0.992, 1.034] lag 2 RR 1.033 [1.013, 1.054] lag 3 RR 1.026 [1.006, 1.047] lag 4 Multipollutant model: NO_2/SO_2 1.031 [1.004, 1.059] lag 3 NO_2/PM_{10} 1.001 [0.971, 1.031] lag 3 NO_2/Pollen 1.024 [1.004, 1.044] lag 3 NO_2/Pollen/O_3 1.024 [1.005, 1.045] Poisson NO_2/Pollen/O_3 1.022 [1.005, 1.040] GAM

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Tenías et al. (1998) Valencia, Spain Period of Study: 1993-1995 Seasons: Cold: Nov-Apr Warm: May-Oct	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: >14 Study Design: Time series N: 734 Statistical Analyses: Poisson regression, APHEA protocol Covariates: seasonality, temperature, humidity, long-term trend, day of wk, holidays, influenza Seasons: Cold: Nov-Apr; Warm: May-Oct Dose-Response Investigated: Yes Statistical Package: NR Lag: 0-3 days	24 h: 57.7 μg/m^3 Cold: 55.9 Warm: 59.4 1-h max: 101.1 μg/m^3 Cold: 97.3 Warm: 102.8 # of Stations: 2	24 h: O_3; r = −0.304 SO_2 (24 h); r = 0.265 SO_2 (1 h); r = 0.261 1 h: O_3; r = −0.192 SO_2 (24 h); r = 0.199 SO_2 (1 h); r = 0.201	Increment: 10 μg/m^3 NO_2 24-h avg All yr 1.076 [1.020, 1.134] lag 0 Cold 1.083 [1.022, 1.148] lag 0 Warm 1.066 [0.989, 1.149] lag 0 NO_2 1-h max All yr 1.037 [1.008, 1.066] lag 0 Cold 1.034 [1.004, 1.066] lag 0 Warm 1.044 [1.002, 1.088] lag 0

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Tenías et al. (2002) Valencia, Spain Period of Study: 1994-1995	Outcome(s): COPD ICD-9 Code(s): NR Age groups analyzed: >14 Study Design: Time series N: 1,298 # of Hospitals: 1 Statistical Analyses: Poisson regression, APHEA protocol; basal models and GAM Covariates: seasonality, annual cycles, temperature, humidity, day of wk, feast days Seasons: Cold, Nov-Apr; Warm, May-Oct Dose-Response Investigated: Yes Statistical Package: NR Lag: 0-3 days	NO_2 24-h avg: 57.7 μg/m^3; Range: 12, 135 1-h max: 100.1 μg/m^3; Range: 31, 305 # of Stations: 6 manual and 5 automatic; r = 0.87	BS; r = 0.246 SO_2; r = 0.194 CO; r = 0.180 O_3; r = −0.192	Increment: 10 μg/m^3 24-h avg NO_2 All Year RR 0.979 [0.943, 1.042] lag 0 Cold, 24-h avg: RR 0.991 [0.953, 1.030] lag 0 Warm, 24-h avg: RR 0.961 [0.900, 1.023] lag 0 1-h max NO_2 All Year RR 0.986 [0.966, 1.007] lag 0 Cold, 24-h avg: RR 0.996 [0.975, 1.018] lag 0 Warm, 24-h avg: RR 0.968 [0.935, 1.003] lag 0 Possibility of a linear relationship between pollution and risk of emergency cases could not be ruled out.

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Migliaretti et al. (2005) Turin, Italy Period of Study: 1997-1999	Outcome (ICD-9): Asthma (493) Age groups analyzed: <15, 15-64, >64 Study Design: Case-Control Controls: age matched with other respiratory disease (ICD-9: 460-487, 490-2, 494-6, 500-19) or heart disease (ICD-9: 390-405, 410-429) N: cases = 1,401 controls = 201,071 Statistical Analyses: logistic regression Covariates: seasonality, temperature, humidity, solar radiation, wind velocity, day of wk, holiday, gender, age, education level Seasons: Cold: Oct-Mar; Warm: Apr-Sep Statistical Package: NR Lag: 0-3 days and cumulative	All Participants: 24-h mean: 112.7 µg/m^3, SD = 30.2, Median = 107.7 Cases: 24-h mean: 117.1 µg/m^3, SD = 30.0, Median = 113.0 Controls: 24-h mean: 112.7 µg/m^3, SD = 30.2, Median = 107.7 # of Stations: 10, r = 0.79	TSP; r = 0.8 Two-pollutant model adjusted for TSP	Increment: 10 µg/m^3 Single Pollutant (NO_2): <15 yrs 2.3% [0.3, 4.40] 15-64 yrs 3.10% [−0.01, 7.70] >64 yrs 7.70% [0.20, 15.20] All ages 2.40% [0.5, 4.30] Copollutant (NO_2 and TSP) <15 yrs 1.71% [−0.02, 5.00] 15-64 yrs 1.20% [−0.06, 6.50] >64 yrs 0.91% [−0.08, 5.91] All ages 1.10% [−0.02, 3.82]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
EUROPE and MIDDLE EAST (cont'd)				
Pantazopoulou et al. (1995) Athens, Greece Period of Study : 1988	Outcomes: All respiratory visits ICD-9: NR Age groups analyzed: All ages Study Design: Time series N: 213,316 Number of hospitals: 14 Statistical Analyses: Multiple linear regression Covariates: Season, day of wk, holiday, temperature, relative humidity Season: Warm (3/22-9/21), Cold (1/1-3/21 and 9/22-12/31) Lag: NR	NO_2 24-h avg Winter: 94 μg/m^3, SD = 25 5th: 59, 50th: 93, 95th: 135 Summer: 111 μg/m^3, SD = 32 5th: 65, 50th: 108, 95th: 173 # of stations: 2	CO BS	Increment: 76 μg/m^3 in winter and 108 μg/m^3 in summer (95th-5th) Respiratory disease admissions Winter: Percent increase: β = 66.8 [19.6, 113.9] Summer: Percent increase: β = 21.2 [−35.1, 77.5]
Garty et al. (1998) Tel Aviv, Israel 1993	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: 1-18 Study Design: Descriptive study with correlations N: 1,076 Statistical Analyses: Pearson correlation and partial correlation coefficients Covariates: maximum and minimum ambient temperatures, relative humidity and barometric pressure Statistical Package: Statistix	24-h mean of NO_x (estimated from histogram): 60 μg/m^3; Range 50, 250		Correlation between NO_x and ER visits for asthma: All Year: Daily data r = 0.30 Running mean for 7 days r = 0.62 Excluding September: Daily data r = 0.37 Running mean for 7 days r = 0.74 38% of variance in number of ER visits explained by fluctuations in NO_x. Increases to 55% when Sept. is omitted from analyses.

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
LATIN AMERICA				
Farhat* et al. (2005) São Paulo, Brazil Period of Study: 1996-1997	Outcome(s) (ICD-9): Lower Respiratory Disease (466, 480-5) Age groups analyzed: <13 Study Design: Time series N: 4,534 # of Hospitals: 1 Statistical Analyses: 1) Poisson regression and 2) GAM - no mention of more stringent criteria Covariates: long-term trends, seasonality, temperature, humidity Statistical Package: S-Plus Lag: 0-7 days, 2,3,4 day moving avg	Mean: 125.3 μg/m^3 SD = 51.7 IQR: 65.04 μg/m^3 # of Stations: 6	PM_{10}; r = 0.83 SO_2; r = 0.66 CO; r = 0.59	Increment: IQR of 65.04 μg/m^3 Single-pollutant models (estimated from graphs): LRD ~17.5% [12.5, 24] Multipollutant models: Adjusted for: PM_{10} 16.1% [5.4, 26.8] 4 day avg SO_2 24.7% [18.2, 31.3] 4 day avg CO 19.2% [11.8, 26.6] 4 day avg Multipollutant model 18.4% [3.4, 33.5] 4 day avg

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
LATIN AMERICA (cont'd)				
Martins* et al. (2002) São Paulo, Brazil Period of Study: 5/96-9/98	Outcome(s) (ICD-10): Chronic Lower Respiratory Disease (CLRD) (J40-J47); includes chronic bronchitis, emphysema, other COPDs, asthma, bronchiectasia Age groups analyzed: >64 Study Design: Time series N: 712 # of Hospitals: 1 Catchment area: 13,163 total ER visits Statistical Analyses: Poisson regression and GAM - no mention of more stringent criteria Covariates: weekdays, time, minimum temperature, relative humidity, daily number of non-respiratory emergency room visits made by elderly Statistical Package: S-Plus Lag: 2-7 days and 3 day moving avgs	NO_2 max 1-h avg ($\mu g/m^3$): 117.6, SD = 53.0, Range 32.1, 421.6 IQR: 62.2 $\mu g/m^3$ # of Stations: 4	O_3; r = 0.44 SO_2; r = 0.67 PM_{10}; r = 0.83 CO; r = 0.62	Increment: IQR of 62.2 $\mu g/m^3$ Percent increase: 4.5% [−6.5, 15] lag 3 day moving avg (estimated from graph)

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
LATIN AMERICA (cont'd)				
Ilabaca et al. (1999) Santiago, Chile Period of Study: 2/1/95 – 8/31/96 Days: 578	Outcome(s) (ICD-9): Upper respiratory illness (460-465, 487); Lower respiratory illness (466, 480-486, 490-494, 496, 519.1, 033.9); Pneumonia (480-486) Age groups analyzed: <15 Study Design: Time series # of Hospitals: 1 Statistical Analyses: Poisson regression Covariates: Long-term trend, season, day of wk, temperature, humidity, influenza epidemic Season: Warm (Sep-Apr), Cool (May-Aug) Statistical Package: NR Lag: 0-3 days	24-h avg NO_2: Warm: Mean: 97.0 Median: 91.5 SD: 34.6 Range: 37.2, 246 5th: 54.3 95th: 163.0 Cool: Mean: 160.2 Median: 154.4 SD: 59.5 Range: 60.1, 397.5 5th: 74.4 95th: 266.0 # of stations: 4, r = 0.70, 0.88	Warm: SO_2 r = 0.66 O_3 r = 0.15 PM_{10} r = 0.71 $PM_{2.5}$ r = 0.70 Cool: SO_2 r = 0.74 O_3 r = 0.22 PM_{10} r = 0.82 $PM_{2.5}$ r = 0.80	Increment: IQR All respiratory Cool Lag 2 IQR: 56.4 RR 1.0378 [1.0211, 1.0549] Lag 3 IQR: 56.4 RR 1.0294 [1.0131, 1.0460] Lag avg 7 IQR: 33.84 RR 1.0161 [1.0000, 1.0325] Warm Lag 2 IQR: 30.08 RR 1.0208 [0.9992, 1.0428] Lag 3 IQR: 30.08 RR 1.0395 [1.0181, 1.0612] Lag avg 7 IQR: 22.56 RR 1.0251 [0.9964, 1.0548] Upper respiratory Cool Lag 2 IQR: 56.4 RR 1.0569 [1.0339, 1.0803] Lag 3 IQR: 56.4 RR 1.0318 [1.0095, 1.0545] Lag avg 7 IQR: 33.84 RR 1.0177 [0.9960, 1.0399] Warm Lag 2 IQR: 30.08 RR 1.0150 [0.9881, 1.0426] Lag 3 IQR: 30.08 RR 1.0425 [1.0157, 1.0699] Lag avg 7 IQR: 22.56 RR 0.9944 [0.9591, 1.0311] Pneumonia Cool Lag 2 IQR: 56.4 RR 1.0824 [1.0300, 1.1374] Lag 3 IQR: 56.4 RR 1.0768 [1.0273, 1.1287] Lag avg 7 IQR: 33.84 RR 1.0564 [1.0062, 1.1092] Warm Lag 2 IQR: 30.08 RR 1.1232 [1.0450, 1.2072] Lag 3 IQR: 30.08 RR 1.0029 [0.9332, 1.0779] Lag avg 7 IQR: 22.56 RR 1.1084 [1.0071, 1.2200]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
LATIN AMERICA (cont'd)				
Lin et al. (1999) São Paulo, Brazil Period of Study: May 1991 - April 1993 Days: 621	Outcome(s): Respiratory disease, Upper respiratory illness, Lower respiratory illness, Wheezing ICD-9 Code(s): NR Age groups analyzed: <13 Study Design: Time series # of Hospitals: 1 Statistical Analyses: Gaussian and Poisson regression Covariates: Long-term trend, seasonality, day of wk, temperature, humidity Statistical Package: NR Lag: 5-day lagged moving avgs	NO_2 µg/m^3: Mean: 163 SD: 85 Range: 2, 688 Number of stations: 3	SO_2 r = 0.38 CO r = 0.35 PM_{10} r = 0.40 O_3 r = 0.15	Increment: NR All respiratory illness NO_2 alone RR 1.003 [1.001, 1.005] 5-day moving avg $NO_2 + PM_{10} + O_3 + SO_2 + CO$ RR 0.996 [0.994, 0.998] Lower respiratory illness NO_2 alone RR 0.999 [0.991, 1.007] 5-day moving avg $NO_2 + PM_{10} + O_3 + SO_2 + CO$ RR 0.990 [0.982, 0.998] Upper respiratory illness NO_2 alone RR 1.003 [0.999, 1.007] 5-day moving avg $NO_2 + PM_{10} + O_3 + SO_2 + CO$ RR 0.996 [0.992, 1.000] Wheezing NO_2 alone RR 0.996 [0.990, 1.002] 5-day moving avg $NO_2 + PM_{10} + O_3 + SO_2 + CO$ RR 0.991 [0.983, 0.999]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
ASIA				
Chew et al. (1999) Singapore Period of Study: 1990-1994	Outcome(s) (ICD-9): Asthma (493) Age groups analyzed: 3-12, 13-21 Study Design: Time series N: 23,000 # of Hospitals: 2 Statistical Analyses: Linear regression, GLM Covariates: variables that were significantly associated with ER visits were retained in the model Statistical Package: SAS/STAT, SAS/ETS 6.08 Lag: 1,2 days avgs	24-h avg: 18.9 µg/m^3, SD = 15.0, Max < 40 # of Stations: 15	SO_2; r = −0.22 O_3; r = 0.17 TSP; r = 0.23	Categorical analysis (via ANOVA) p-value and Pearson correlation coefficient (r) using continuous data comparing daily air pollutant levels and daily number of ER visits Age Group: 3-12 13-21 Lag 0 r = 0.10 r = 0.09 p = 0.0019 p < 0.001 Lag 1 r = 0.12 r = 0.04 p < 0.001 p = 0.0014 Lag 2 r = 0.14 r = 0.03 p < 0.001 p = 0.0066
Hwang and Chan (2002) Taiwan Period of Study: 1998	Outcome(s) (ICD-9): Lower Respiratory Disease (LRD) (466, 480-6) including acute bronchitis, acute bronchiolits, pneumonia Age groups analyzed: 0-14, 15-64, ≥65, all ages Study Design: Time series Catchment area: Clinic records from 50 communities Statistical Analyses: Linear regression, GLM Covariates: temperature, dew point temperature, season, day of wk, holiday Statistical Package: NR Lag: 0,1,2 days and avgs	24-hr avg: 23.6 ppb, SD = 5.4, Range: 13.0, 34.1	SO_2 PM_{10} O_3 CO No correlations for individual pollutants. Colinearity of pollutants prevented use of multipollutant models	Increment: 10% change in NO_2 (natural avg) which is equivalent to 2.4 ppb. NOTE: The percent change is for the rate of clinic use NOT for relative risk for adverse effect. Increased clinic visits for lower respiratory disease (LRD) by age group 0-14 yrs 1.3% [1.0, 1.6] lag 0 15-64 yrs 1.5% [1.3, 1.8] lag 0 ≥65 yrs 1.8% [1.4, 2.2] lag 0 All ages 1.4% [1.2, 1.6] lag 0

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: EMERGENCY DEPARTMENT VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels of NO_2 & Monitoring Stations	Copollutant Correlations	Effects and Interpretation: Relative Risk & Confidence Intervals (95%)
ASIA (cont'd)				
Tanaka et al. (1998) Kushiro, Japan Period of Study: 1992-1993	Outcome(s): Asthma ICD-9 Code(s): NR Age groups analyzed: 15-79 Study Design: Time series N: 102 # of Hospitals: 1 Statistical Analyses: Poisson regression Covariates: temperature, vapor pressure, barometric pressure, relative humidity, wind velocity, wind direction at maximal velocity Statistical Package: NR	NO_2 24-h avg 9.2 ± 4.6 ppb in fog 11.5 ± 5.7 in fog free days Max NO_2 24-h avg <30 ppb	NO_2; r = NR SO_2; r = NR SPM (TSP); r = 0.70 O_3; r = NR	Increment: 15 ppb Nonatopic OR 0.62 [0.45, 0.84] Atopic OR 0.81 [0.67, 0.97]

*Default GAM
+Did not report correction for over-dispersion
NR: Not Reported
APHEA: Air Pollution and Health: a European Approach

TABLE AX6.3-3. RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: GENERAL PRACTITIONER/CLINIC VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants Correlations	Effects: Relative Risk & Confidence Intervals (95%)
NORTH AMERICA				
Hernández-Garduño et al. (1997) Mexico City, Mexico Period of Study: May 15, 1992 - January 31, 1993	Outcome(s): Respiratory illness ICD9: NR Age groups analyzed: <15, 15+, all ages (0-92) Study Design: Time series N: 24,113 Number of Clinics: 5 Statistical Analyses: Cross-correlation, linear regression and autoregressive error model analyses Covariates: Long-term trend, day of wk, temperature, humidity Statistical Package: SAS Lag: 0-6	Number of Stations: 5	O_3 SO_2 CO NO_x	Increment: Maximum NO_2 concentration of all days-Mean NO_2 concentration of all days <14 yrs NO_2 lag 0: RR 1.29 ± 0.09 ($p < 0.01$) NO_2 lag 6: RR 1.18 ± 0.09 ($p > 0.05$) 15+ yrs NO_2 lag 0: RR 1.14 ± 0.07 ($p < 0.05$) NO_2 lag 6: RR 1.10 ± 0.06 ($p > 0.05$) All ages NO_2 lag 0: RR 1.43 ± 0.15 ($p < 0.01$) NO_2 lag 6: RR 1.29 ± 0.15 ($p > 0.05$)
CANADA				
Villeneuve et al. (2006) Toronto, ON, Canada Period of Study: 1995-2000 Days: 2,190	Outcome(s) (ICD9): Allergic Rhinitis (177) Age groups analyzed: ≥65 Study Design: Time series N: 52,691 Statistical Analyses: GLM, using natural splines (more stringent criteria than default) Covariates: Day of wk, holiday, temperature, relative humidity, aero-allergens Season: All yr; Warm, May-Oct; Cool, Nov-Apr Statistical Package: S-Plus Lag: 0-6	24-h avg: 25.4 ppb, SD = 7.7 IQR: 10.3 ppb, range 9.2, 71.7 Number of stations: 9	SO_2 O_3 CO $PM_{2.5}$ $PM_{10-2.5}$ PM_{10}	Increment: 10.3 ppb (IQR) All results estimated from Stick Graph: All Yr: Mean Increase: 1.9% [−0.2, 3.8] lag 0 Warm: Mean Increase: 0.1% [−3.2, 3.8] lag 0 Cool: Mean Increase: 1.4% [0.0, 5.9] lag 0

TABLE AX6.3-3 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: GENERAL PRACTITIONER/CLINIC VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants Correlations	Effects: Relative Risk & Confidence Intervals (95%)
EUROPE				
Hajat et al. (1999) London, United Kingdom Period of Study: 1992-1994	Outcome (ICD9): Asthma (493); Lower respiratory disease (464, 466, 476, 480-3, 490-2, 485-7, 4994-6, 500, 503-5, 510-5) Age groups analyzed: 0-14; 15-64; 65+; all ages Study Design: Time series analysis Statistical Analysis: Poisson regression, APHEA protocol Covariates: long-term trends, seasonality, day of wk, temperature, humidity Seasons: Warm, Apr-Sep; Cool, Oct-Mar; All yr Dose-response investigated? Yes Statistical package: SAS Lag: 0-3 days, cumulative	All yr 24-h avg: 33.6 ppb, SD = 10.5 Warm: 32.8 (19.8) Cool: 34.5 (10.1) 10th-90th all yr percentile: 24 ppb	SO_2; r = 0.61 BS; r = 0.70 CO; r = 0.72 PM_{10}; r = 0.73 O_3; r = −0.10	Increment: 24 ppb (90th-10th percentile) Asthma All ages 2.1% [−0.7, 4.9] lag 0; 3.1% [−0.4, 6.7] lag 0-1 0-14 yrs 6.1% [1.2, 11.3] lag 1; 6.9% [1.7, 12.4] lag 0-1 Warm: 13.2% [5.6, 21.3] lag 1 Cool: −0.1% [−6.3, 6.5] lag 1 15-64 yrs 3.0% [−0.7, 6.7] lag 0; 3.1% [−1.6, 7.9] lag 0-3 Warm: 3.3% [−2.0, 8.9] lag 0 Cool: 2.6% [−2.3, 7.7] lag 0 65+ yrs 9.9% [1.6, 18.7] lag 2; 5.3% [−3, 14.3] lag 0-3 Warm: 18.6% [6.3, 32.4] lag 2 Cool: −0.5% −9.6, 11.8] lag 2 Lower Respiratory disease All ages 1.3% [−0.4, 3.0] lag 1; 1.2% [−0.7, 3.1] lag 0-2 0-14 yrs 4.8% [1.3, 8.3] lag 2; 4.5% [0.4, 8.7] lag 0-3 Warm: 1.4% [−3.8, 6.9] lag 2 Cool: 7.2% [2.8, 11.6] lag 2 15-64 yrs 1.1% [−1.1, 3.4] lag 2; 0.8% [−1.8, 3.5] lag 0-2 Warm: 2.3% [−1.2, 5.9] lag 2 Cool: 0.2% [−2.6, 3.1] lag 2 65+ −1.7% [−4.3, 1.1] lag 0 Warm: −1.7% [−5.9, 2.6] lag 0 Cool: −1.6% [−4.8, 1.8] lag 0 Two-pollutant model-Asthma NO_2 alone 5.2% [0.8, 9.8] NO_2/O_3 6.7% [2.2, 11.4] NO_2/SO_2 3.9% [−1.2, 9.2] NO_2/PM10 5.3% [−0.6, 11.6] Two-pollutant model - Lower Respiratory disease NO_2 alone 4.2% [1.1, 7.3] NO_2/O_3 4.9% [1.8, 8.2] NO_2/SO_2 2.5% [−1.1, 6.2] NO_2/PM_{10} 3.5% [0.1, 6.9]

TABLE AX6.3-3 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: GENERAL PRACTITIONER/CLINIC VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants Correlations	Effects: Relative Risk & Confidence Intervals (95%)
EUROPE (cont'd)				
Hajat* et al. (2001) London, United Kingdom Period of Study: 1992-1994	Outcome (ICD9): Allergic Rhinitis (477) Age groups analyzed: 0-14; 15-64; 65+; all ages Study Design: Time series analysis N: 4,214 Statistical Analysis: Poisson regression, GAM Covariates: long-term trends, seasonality, day of wk, temperature, humidity, variation in practice population, counts for lagged allergic pollen measures, daily number of consultations for influenza Dose-response investigated? Yes Statistical package: S-Plus Lag: 0-6 days, cumulative	NO_2 24-h avg: 33.6 ppb, SD = 10.5 # of Stations: 3; r = 0.7-0.96	SO_2; r = 0.61 BS; r = 0.70 CO; r = 0.72 PM_{10}; r = 0.73 O_3; r = −0.10	Increment: 24 ppb (90th-10th percentile) Single-pollutant model <1 to 14 yrs 11.0% [3.8, 18.8] lag 4 12.6% [4.6, 21.3] lag 0-4 15 to 64 yrs 5.5% [2.0, 9.1] lag 6 11.1% [6.8, 15.6] lag 0-6 >64 yrs - too small for analysis Two-pollutant models <1 to 14 yrs NO_2 & O_3: 7.9% [0.6, 15.8] NO_2 & SO_2: −3.8% [11.8, 5.0] NO_2 & PM_{10}: 10.8% [0.1, 22.7] 15 to 64 yrs NO_2 & O_3: 4.8% [1.0, 8.8] NO_2 & SO_2: 1.0% [−3.7, 5.8] NO_2 & PM_{10}: 0.5% [−4.9, 6.3]

TABLE AX6.3-2 (cont'd). RESPIRATORY HEALTH EFFECTS OF OXIDES OF NITROGEN: GENERAL PRACTITIONER/CLINIC VISITS

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants Correlations	Effects: Relative Risk & Confidence Intervals (95%)
EUROPE (cont'd)				
Hajat* et al. (2002) London, United Kingdom Period of Study: 1992-1994	Outcome (ICD9): Upper Respiratory Disease, excluding Rhinitis (460-3, 465, 470-5, 478) Age groups analyzed: 0-14; 15-64; 65+; all ages Study Design: Time series analysis Statistical Analysis: Poisson regression, GAM Covariates: long term trends, seasonality, day of wk, holidays, temperature, humidity, variation in practice population, counts for lagged allergic pollen measures, daily number of consultations for influenza Seasons: Warm, Apr-Sep; Cool Oct-Mar Dose-response investigated? Yes Statistical package: S-Plus Lag: 0,1,2,3 days	NO_2 24 h avg: 33.6 ppb, SD = 10.5 Warm (April-Sept) Mean: 32.8 ppb, SD = 10.1 Cool (Oct-March) Mean: 34.5 ppb, SD = 10.1 # of Stations: 3	SO_2; r = 0.61 BS; r = 0.70 CO; r = 0.72 PM_{10}; r = 0.73 O_3; r = −0.10	Increment (90th-10th percentile): All yr: 24 ppb; Warm season: 25.8 ppb; Cool season: 22.1 ppb Single-pollutant model All yr 0-14 yr 2.0% [−0.3, 4.3] lag 3 15-64 yrs 5.1% [2.0, 8.3] lag 2 >65 yrs 8.7% [3.8, 13.8] lag 2 Warm 0-14 yrs 2.5% [−0.9, 6.1] lag 3 15-64 yrs 6.7% [3.7, 9.8] lag 2 ≥65 yrs 6.6% [−1.1, 14.9] lag 2 Cool 0-14 yrs 1.7% [−1.1, 4.6] lag 3 15-64 yrs 1.2% [−1.3, 3.9] lag 2 >65 yrs 9.4% [2.8, 16.4] lag 2 Two-pollutant models 0-14 yrs NO_2 & O_3: 1.7% [−0.6, 3.9] NO_2 & SO_2: 2.2% [−0.4, 5.0] NO_2 & PM_{10}: 1.5% [−1.7, 4.8] For 15-64 yrs NO_2 & O_3: 4.4% [2.2, 6.8] NO_2 & SO_2: 4.4% [1.6, 7.2] NO_2 & PM_{10}: 2.7% [−0.5, 5.9] For >65 yrs NO_2 & O_3: 8.1% [3.0, 13.6] NO_2 & SO_2: 8.6% [2.1, 15.4] NO_2 & PM_{10}: 4.3% [−2.8, 11.8]

* Default GAM

\+ Did not report correction for over-dispersion

NR: Not Reported

APHEA: Air Pollution and Health: a European Approach

TABLE AX6.4-1. HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Burnett et al. (1997) * Metropolitan Toronto (Toronto, North York, East York, Etobicoke, Scarborough, York), Canada Study period: 1992-1994, 388 days, summers only	Outcome(s) (ICD9): IHD 410-414; Cardiac Dysrhythmias 427; Heart failure 428. All Cardiac 410-414, 427, 428. Obtained from hospital discharge data. Population: 2.6 Million residents Study design: Time series Age groups analyzed: all # Hospitals: NR Statistical analysis: relative risk regression models, GAMs. Covariates: adjusted for long-term trends, seasonal and subseasonal variation, day of the wk, temperature, dew point Seasons: summer only Dose response: Figures presented Statistical package: NR Lag: 1-4 days	NO_2 daily 1-h max (ppb): Mean: 38.5 CV: 29 Min: 12 25th percentile: 31 50th percentile: 38 75th percentile: 45 Max: 81 # of Stations: 6-11 (Results are reported for additional metrics including 24 h avg and daytime avg (day))	H^+ (0.25) SO_4 (0.34) TP (0.61) FP (0.45) CP (0.61) COH (0.61) O_3 (0.07) SO_2 (0.46) CO (0.25)	Results reported for RR for an IQR increment increase in NO_2. T ratio in parentheses. All Cardiac Disease Single-pollutant model 1.049 (3.13), daily avg over 4 days, lag 0 Multipollutant model 1.30 (1.68), w/ NO_2, O_3, SO_2, Objective of study was to evaluate the role of particle size and chemistry on cardio and respiratory diseases. NO_2 attenuated the effect of particulate in this study.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Burnett et al. (1999) * Metropolitan Toronto (Toronto, North York, East York, Etobicoke, Scarborough, York), Canada Study Period: 1980-1995, 15 yr	Outcome(s) (ICD9): IHD 410-414; Cardiac Dysrhythmias 427; Heart failure 428; All Cardiac 410-414, 427, 428; Cerebrovascular Disease Obtained from hospital discharge data 430-438; Peripheral Circulation Disease 440-459. Population: 2.13-2.42 million residents Study Design: Time series Statistical Analysis: GAMs to estimate log RR per unit changes, stepwise regression used to select minimum number of air pollutants in multipollutant models. Covariates: long-term trends, seasonal variation, day of wk, temperature, and humidity. Statistical Package: SPLUS Lag(s): 0-2 day	NO_2 daily avg (ppb) Mean: 25.2 5th percentile: 13 25th percentile: 19 50th percentile: 24 75th percentile: 30 95th percentile: 42 Max: 82 Multiple day avgs used in models	$PM_{2.5}$ (0.50) $PM_{10-2.5}$ (0.38) PM_{10} (0.52) CO (0.55) SO_2 (0.55) O_3 (−0.04)	Results reported for % increase in hospital admissions for an increment increase in NO_2 equal to the mean value. Single Pollutant Models: Dysrhythmias: 5.33 (1.73) 3-day avg, lag 0 Heart Failure: 9.48 (6.33), 1 day, lag 0 IHD: 9.73 (8.4) 2-day avg, lag 0 Cerebrovascular disease: 1.98 (1.34), 1 day, lag 0 Peripheral circulation: 3.57 (1.78), 1-day, lag 0 Multipollutant Models: Heart failure 6.89 (w/ CO) 6.68 (w/ CO, $PM_{2.5}$) 6.33 (w/ CO, $PM_{2.5}$, $PM_{10-2.5}$) 6.45 (w/ CO, $PM_{2.5}$, $PM_{10-2.5}$, PM_{10}) IHD 8.34 (w/ CO, SO_2) 7.76 (w/ CO, SO_2, $PM_{2.5}$) 8.41 (w/ CO, SO_2, $PM_{2.5}$, $PM_{10-2.5}$) 8.52 (w/ CO, SO_2, $PM_{2.5}$, $PM_{10-2.5}$, PM_{10}) In multipollutant models, gaseous pollutants were selected by stepwise regression. PM variables were then added to the model.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Morris et al. (1995) US (Chicago, Detroit, LA, Milwaukee, NYC, Philadelphia) Study Period: 1986-1989, 4 yr	Outcome(s) (ICD9): CHF 428. Daily Medicare hospital admission records. Study Design: Time series Statistical Analyses: GLM, negative binomial distribution Age groups analyzed: ≥65 yrs Covariates: temperature, indicator variables for mo to adjust for weather effects and seasonal trends, day of wk, yr Statistical Software: S-PLUS Lag(s): 0-7 day	NO_2 1 h-max (ppb) Mean (SD) LA: 0.077 (0.028) Chicago: 0.045 (0.013) Philadelphia: 0.054 (0.017) New York: 0.064 (0.022) Detroit: 0.041 (0.015) Houston: 0.041 (0.017) Milwaukee: 0.040 (0.014)	SO_2 1-h max O_3 1-h max CO 1-h max Correlations of NO_2 with other pollutants strong. Multipollutant models run.	Results reported for RR of admission for CHF associated with an incremental increase in NO_2 of 10 ppb. CHF: LA: 1.15 (1.10, 1.19) Chicago: 1.17 (1.07, 1.27) Philadelphia: 1.03 (0.95, 1.12) New York: 1.07 (1.02, 1.13) Detroit: 1.04 (0.92, 1.18) Houston: 0.99 (0.88, 1.10) Milwaukee: 1.05 (0.89, 1.23) RR diminished in multipollutant models (4 copollutants).

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Wellenius et al. (2005a) Birmingham, Chicago, Cleveland, Detroit, Minneapolis, New Haven, Pittsburgh, Seattle Study Period: Jan 1986-Nov 1999 (varies slightly depending on city)	Outcome(s) IS, primary diagnosis of acute but ill-defined cerebrovascular disease or occlusion of the cerebral arteries; HS, primary diagnosis of intracerebral hemorrhage. ICD codes not provided. Hospital admissions ascertained from the Centers for Medicare and Medicaid Services. Cases determined from discharge data were admitted from the ER to the hospital. N IS: 155,503 N HS: 19,314 Study Design: Time-stratified case crossover. Control days chosen such that they fell in same mo and same day of wk. Design controls for seasonality, time trends, chronic and other slowly varying potential confounders. Statistical Analysis: 2-stage hierarchical model (random effects), conditional logistic regression for city effects in the first stage Software package: SAS Covariates: Lag(s): 0-2, unconstrained distributed lags	NO_2 24 h (ppb) 10th: 13.71 25th: 18.05 Median: 23.54 75th: 29.98 90th: 36.54 NO_2 data not available for Birmingham, Salt Lake, and Seattle	PM_{10} (0.53) CO, SO_2 Correlation only provided for PM because study hypothesis involves PM	Results reported for percent increase in stroke admissions for an incremental increase in NO_2 equivalent to one IQR (11.93). Ischemic Stroke: 2.94 (1.78, 4.12), lag 0 Hemorrhagic Stroke: 0.38 (−2.66, 3.51), lag 0 Multipollutant models not run.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Fung et al. (2005) Windsor, Ontario, Canada Study Period: April 1995-Jan 2000	Outcome(s) (ICD9): CHF 428; IHD 410-414; dysrhythmias 427. Hospital admissions from Ontario Health Insurance Plan records. Study Design: Time series Statistical Analysis: GLM N: 11,632 cardiac admission, 4.4/day for 65+ age group Age groups analyzed: 65+, <65 yr Statistical Software: SPLUS Lag(s): lag 0, 2, 3 day avg	NO_2 1-h max (ppb): Mean (SD): 38.9 (12.3) Min: 0 Max: 117	SO_2 (0.22) CO (0.38) O_3 (0.26) COH (0.39) PM_{10} (0.33)	Results expressed as percent change associated with an incremental increase in NO_2 equivalent to the IQR (16 ppb) Cardiac: 65+ age group: 0.8 (2.2, 3.9), lag 0 0.9 (−2.7, 4.6), 2-day avg (lag 0-1) 0.8 (−3.3, 5.0), 3-day avg (lag 0-2) Effect for NO_2 not observed in these data. Association of SO_2 with cardiac admissions observed.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Linn et al. (2000) * Metropolitan Los Angeles, USA Study Period: 1992-1995	Outcome(s) (ICD9): CVD 390-459; Cerebrovascular 430-438; CHF 428; MI 410; cardiac ARR 427; Occlusive Stroke 430-435. Hospital admission records used to ascertain cases. Study Design: Time series Statistical Analyses: Poisson regression, GAM Covariates: day of wk, holidays, long-term trend, seasonal variation, temperature, humidity Lag(s): 0-1 Seasons: Winter, Spring, Summer, Autumn Statistical Software: SPSS, SAS	NO_2 24 h (pphm) Winter Mean: (SD) 3.4 (1.3) Range: 1.1, 9.1 Spring Mean (SD): 2.8 (0.9) Range: 1.1, 6.1 Summer Mean (SD): 3.4 (1.0) Range: 0.7, 6.7 Autumn Mean (SD): 4.1 (1.4) Range: 1.6, 8.4	CO (0.84, 0.92) O_3 (−0.23, 0.11) PM_{10} (−0.67, 0.8) Range in correlations depends on the season, independent effects of pollutants could not be distinguished. # Stations: 6+	Results reported as increase % increase in admission for a 10 ppb increase in NO_2. SD in parentheses. Season-specific increases reported when statistically significant. CVD All Seasons: 1.4 (0.2) Winter: 1.6 (0.4) Spring: 0.1 (0.6) Summer: 1.1 (0.5) Autumn: 1.4 (0.3) Cerebrovascular All Seasons: 0.4 (0.4) Winter: −1.3 (0.7) Spring: 4.2 (1.2) Summer: 0.9 (1.2) Autumn: 0.7 (0.6) MI 1.1 (0.5) CHF 1.0 (0.5), winter 1.9 (0.9) Cardiac Arrhythmia 0.6 (0.5) Occlusive stroke 2.0 (0.5), winter 2.7 (1.0), autumn 0.1 (0.05)

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Lippmann et al. (2000*; reanalysis Ito, 2003, 2004) Windsor Ontario (near Detroit MI) Study period: 1992-1994 (hospital admissions – mortality study spanned longer period)	Outcome(s): IHD 410-414; dysrhythmias 427; heart failure 428; stroke 431-437. Study Design: Time series Statistical Analysis: Poisson regression GAM. Results of reanalysis by Ito 2003, 2004 with GLM are presented. Lag(s): 0-3 day	NO_2 24-h avg (ppb) 5th %: 11 25th %: 16 50th %: 21 75th %: 26 95th %: 36 Mean: 21.3	PM_{10} (0.49) $PM_{2.5}$ (0.48) $PM_{10-2.5}$ (0.32) H^+ (0.14) SO_4 (0.35) O_3 (0.14) SO_2 (0.53) CO (0.68)	Results reported for RR for incremental increase in NO_2 of 5th to 95th percentile. IHD 1.01 (0.94, 1.10), lag 0 Dysrhythmias 0.98 (0.86, 1.12) Heart Failure 1 (0.91, 1.09) Stroke 0.99 (0.90, 1.09)
Mann et al. (2002)* South coast air basin of CA, US Study Period: 1988-1995, 8 yr	Outcome(s) IHD 410-414; or IHD with accompanying diagnosis of CHF 428; or Arrhythmia 426, 427; Ascertained through health insurance records. Study Design: Time series N: 54,863 IHD admissions Age groups analyzed: ≤40; 40-59; ≥60. Statistical Analysis: Poisson regression with GAM, results pooled across air basins using inverse variance weighting as no evidence of heterogeneity was observed. Covariates: study day, temperature, relative humidity, day of wk. Lag(s): 0-2, 2-4 day moving avg Software: SPLUS Seasons: Some analyses restricted to April-October	NO_2 24-h avg (ppb): Exposure assigned for each air basin based on health insurance participant's zip code. Mean (SD): 37.2 (15.7) Range: 3.69, 138 Median: 34.8 # Stations: 25-35	O_3 8 h-max (−0.16, 0.54) CO 8-h max (0.64, 0.86) PM_{10} 24-h avg (0.36, 0.60) Range depends on air basin No multipollutant models run. Traffic pollution generally implicated in findings.	Results reported for percent increase in admissions for a 10 ppb incremental increase in NO_2. All IHD 1.68 (1.08, 2.28) IHD w/ secondary diagnosis of Arrhythmia: 1.81 (0.78, 2.85) IHD w/ secondary diagnosis of CHF: 2.32 (0.69, 3.98) IHD w/ no secondary diagnosis: 0.46 (−0.81, 1.74) Effect of secondary diagnosis strongest in the 40-59 age group. Group with secondary CHF may be sensitive subpopulation or their vulnerability may be due to greater prevalence of MI as the primary diagnosis.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Metzger et al. (2004) Atlanta, GA Period of Study: Jan 1993-Aug 31 2000, 4 yr	Outcome(s): IHD 410-414; AMI 410; Dysrhythmias 427; cardiac arrest 427.5; congestive heart failure 428; peripheral and cerebrovascular disease 433-437, 440, 443-444, 451-453; atherosclerosis 440; stroke 436. ED visits from billing records. N: 4,407,535 visits, 37 CVD visits/day # Hospitals: 31 Age groups analyzed: adults ≥19, elderly 56+ Statistical Analysis: Poisson regression, GLM. Sensitivity analyses using GEE and GAM (strict convergence criteria) Covariates: long-term trends, mean and dew point temp, relative humidity (cubic splines) Statistical Software: SAS Season: Warm, April 15-October 14; Cool, October 15-April 14. Lag(s): 0-3 day	NO_2 1-h max (ppb): Median: 26.3 10th-90th percentile range 25, 68	PM_{10} 24 h (0.49) O_3 8-h max (0.42) SO_2 (0.34) CO 1 h (0.68) 1998-2000 Only $PM_{2.5}$ (0.46) Course PM (.46) Ultrafine PM (.26) Water-soluble metals (.32) Sulfates (.17) OC (0.63) EC (.37) OHC (0.3) Multipollutant models used. All models specified a priori.	Results presented for RR of an incremental increase in NO_2 equivalent to 1 SD (3-day moving avg). All CVD: 1.025 (1.012, 1.039) Dysrhythmia: 1.019 (0.994, 1.044) CHF: 1.010 (0.981, 1.040) IHD: 1.029 (1.005, 1.053) PERI: 1.041 (1.013, 1.069) Finger wounds 1.010 (0.993, 1.027) NO_2 effect was generally attenuated in two-pollutant models. The attenuation was strongest in the period after 1998.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Moolgavkar (2000b)* Cook County IL, Los Angeles County, CA, Maricopa County, AZ 1987-1995	Outcome(s) (ICD9): CVD 390-429; Cerebrovascular disease 430-448. Hospital admissions from CA department of health database. Age groups analyzed: 20-64, 65+ yrs Study Design: Time series N: 118 CVD admissions/day # Hospitals: NR Statistical Analysis: Poisson regression, GAM Covariates: adjustment for day of wk, long term temporal trends, relative humidity, temperature Statistical Package: SPLUS Lag: 0-5 days	NO_2 24-h avg (ppb) Cook County: Min: 7 Q1: 20 Median: 25 Q3: 30 Max: 58 NO_2 24-h avg (ppb) LA County: Min: 10 Q1: 30 Median: 38 Q3: 48 Max: 102 NO_2 24-h avg (ppb) Maricopa County: Min: 2 Q1: 14 Median: 19 Q3: 26 Max: 56	PM_{10} (0.22, 0.70) $PM_{2.5}$ (0.73) (LA only) CO (0.63, 0.80) SO_2 (0.02, 0.74) O_3 (−0.23, 0.02) Two-pollutant models (see results)	Results reported for percent change in hospital admissions per 10 ppb increase in NO_2. T statistic in parentheses. CVD, 65+: Cook County 2.9 (10.2), lag 0 2.3 (6.7), lag 0, two-pollutant model (PM_{10}) 2.9 (8.1), lag 0, two-pollutant model (CO) 2.8 (8.8), lag 0, two-pollutant model (SO_2) LA County 2.3 (16.7), lag 0 −0.1 (−0.5), lag 0, two-pollutant model (CO) 1.7 (8.0), lag 0, two-pollutant model (SO_2) Maricopa County 2.9 (4.1), lag 0 −0.3 (−0.3), lag 0, two-pollutant model (CO) 2.6 (3.6), lag 0, two-pollutant model (SO_2) Cerebrovascular Disease, 65+: Cook County 1.6 (3.6) LA County (5.7) Effect size generally diminished with increasing lag time. Increase in hospital admissions (1.3 for CVD and 1.9 for cerebrovascular) also observed for the 20-64 age group.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Moolgavkar (2003) Cook County IL, Los Angeles County, CA, Maricopa County, AZ 1987-1995	Outcome(s) (ICD9): CVD 390-429; Cerebrovascular disease 430-448 was not considered in the reanalysis. Hospital admissions from CA department of health database. Age groups analyzed: 20-64, 65+ yrs Study Design: Time series N: 118 CVD admissions/day # Hospitals: NR Statistical Analysis: Poisson regression, GAM with strict convergence criteria (10-8), GLM using natural splines Covariates: adjustment for day of wk, long-term temporal trends, relative humidity, temperature Statistical Package: SPLUS Lag: 0-5 days	NO_2 24-h avg (ppb) Cook County: Min: 7 Q1: 20 Median: 25 Q3: 30 Max: 58 NO_2 24-h avg (ppb) LA County: Min: 10 Q1: 30 Median: 38 Q3: 48 Max: 102 NO_2 24-h avg (ppb) Maricopa County: Min: 2 Q1: 14 Median: 19 Q3: 26 Max: 56	PM_{10} (0.22, 0.70) $PM_{2.5}$ (0.73) (LA only) CO (0.63, 0.80) SO_2 (0.02, 0.74) O_3 (−0.23, 0.02) Two-pollutant models (see results)	Results for CVD not shown but use of stringent criteria in GAM did not alter results substantially. However, increased smoothing of temporal trends attenuated results for all gases.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Peel et al. (2006) Atlanta, GA Study Period: Jan 1993-Aug 2000	Outcome(s) (ICD9): IHD 410-414; dysrhythmia 427; CHF 428; peripheral vascular and cerebrovascular disease 433-437, 440, 443, 444, 451-453. Computerized billing records for ED visits. Comorbid conditions: hypertension 401-405; diabetes 250; dysrhythmia 427, CHF 428; atherosclerosis 440; COPD 491, 492, 496; pneumonia 480-486; upper respiratory infection 460-465, 466.0; asthma 493, 786.09. # Hospitals: 31 N: 4,407,535 visits Study Design: case crossover. CVD outcomes among susceptible groups with Comorbid conditions. Statistical Analyses: Conditional logistic regression. Covariates: cubic splines for temperature and humidity included in models. Time independent variables controlled through design. Statistical Software: SAS Lag(s): 3-day avg, lagged 0-2 day	NO_2 1-h max (ppb): Mean (SD): 45.9 (17.3) 10th: 25.0 90th: 68.0	PM_{10} 24-h avg O_3 8-h max SO_2 1-h max CO 1-h max Correlations not reported	Results expressed as OR for association of CVD admissions with a 20 ppb incremental increase in NO_2. Comorbid Hypertension IHD: 1.036 (0.997, 1.076) Dysrhythmia: 1.095 (1.030, 1.165) PERI: 1.031 (0.987, 1.076) CHF: 1.037 (0.985, 1.090) Comorbid Diabetes: IHD: 1.003 (0.95, 1.059) Dysrhythmia: 1.158 (1.046, 1.282) PERI: 1.012 (0.947, 1.082) CHF: 1.017 (0.959, 1.078)

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Schwartz, (1997) * Tuscon, AZ Study Period: Jan 1988-Dec 1990.	Outcome(s) (ICD9): CVD 390-429. Ascertained from hospital discharge records. Study Design: Time series Statistical Analysis: Poisson regression, GAM Age groups analyzed: 65+ Covariates: long-term and seasonal trends, day of the wk, temperature, dew point, Statistical Software: SPLUS	NO_2 24-h avg (ppb): Mean: 19.3 10th: 9.9 25th: 13.2 50th: 19 75th: 24.6 90th: 29.8	PM_{10} (0.326) O_3 (−0.456) SO_2 (0.482) CO (0.673)	Results reported as a percent increase in admission for an increment in NO_2 equivalent to the IQR (11.4 ppb). CVD 0.69% (−2.3, 3.8) Tuscon selected to minimize correlations between pollutants. Since there was no association between NO_2 and admissions, author suggests results for CO not confounded by NO_2.
Stieb et al. (2000) * Saint John, New Brunswick Canada Study Period: July 1992-March 1996	Outcome(s): Angina pectoris; MI; dysrhythmia/conduction disturbance; CHF; All Cardiac. ED Visits collected prospectively. Study Design: Time series Statistical Analyses: Poisson regression, GAM N: 19,821 ER visits # Hospitals: 2 Lag(s): 1-8 days	NO_2 24-h avg (ppb) Mean (SD): 8.9 (5.5) 95th: 19 Max: 35 NO_2 max (ppb) Mean (SD): 20.2 95th: 39 Max: 82	CO (0.68) H_2S (−0.07) O_3 (−0.02) SO_2 (0.41) PM_{10} (0.35) $PM_{2.5}$ (0.35) H^+ (−0.25) SO_4 (0.33) COH (0.49)	Results reported for percent change in admissions based on a single pollutant model for incremental increase in NO_2 equivalent to 1 IQR (8.9 ppb) Cardiac visits: −3.9, p-value = 0.136, lag 2, all yr 10.1, p-value = 0.051, lag 5, May-September For specific CVD diagnoses, ARR and CHF approached significance. NO_2 was not a focus of this paper.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Villeneuve et al. (2006) Edmonton, Canada Study Period: April 1992-March 2002	Outcome(s) (ICD9): Acute ischemic stroke 434, 436; hemorrhagic stroke 430, 432; transient ischemic attach (TIA) 435; Other 433, 437, 438. ED visits supplied by Capital Health. N: 12,422 Stroke Visits Catchment area: 1.5 million people Study Design: case-crossover, exposure index time compared to referent time. Time-independent variables controlled in the design. Index and referent day matched by day of wk. Statistical Analysis: Conditional logistic regression, stratified by season and gender. Covariates: temperature and humidity Statistical Software: SAS Season: Warm: April-September; Cool: October-March. Lag(s): 0, 1, 3 day avg	NO_2 24 h ppb: All yr Mean (SD): 24 (9.8) Median: 22.0 25th: 16.5 75th: 30.0 IQR: 13.5 Summer Mean (SD): 18.6 (6.4) Median: 17.5 25th: 14.0 75th: 22.0 IQR: 8 Winter Mean (SD): 29.4 (9.6) Median: 28.5 25th: 22.5 75th: 35.5 IQR: 13.0	O_3 24-h max (−0.33) O_3 24-h avg (−0.51) SO_2 25-h avg (0.42) CO 24-h avg (0.74) PM_{10} 24-h avg (0.34) $PM_{2.5}$ 24-h avg (0.41) All yr correlations summarized.	Results reported for an incremental increase in NO_2 equivalent to one IQR NO_2. Ischemic Stroke, Summer 1.17 (1.05, 1.31), lag 0 1.18 (1.05, 1.31), lag 1 1.26 (1.09, 1.46), 3 day avg Hemorrhagic stroke, Summer 1.16 (0.99, 1.37) 1.14 (0.97, 1.35) 1.18 (0.95, 1.46) TIA not associated with increase in NO_2. Above results are strongest effects, which were observed during summer. Authors attribute NO_2 effect to vehicular traffic since NO_2 and CO are highly correlated.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Wellenius et al. (2005b) Allegheny County, PA (near Pittsburgh) Study Period: Jan 1987-Nov 1999	Outcome(s): CHF 428. Cases are Medicare patients admitted from ER with discharge of CHF. Study Design: Case crossover, control exposures same mo and day of wk, controlling for season by design. Statistical Analysis: Conditional logistic regression N: 55,019 admissions, including repeat admissions, 86% admitted ≤5 times Age groups analyzed: 65+ yrs (Medicare recipients) Covariates: Temperature and pressure. Effect modification by age, gender, secondary diagnosis arrhythmias, atrial fibrillation, COPD, hypertension, type 2 diabetes, AMI within 30 days, angina pectoris, IHD, acute respiratory infection. Statistical Software: SAS Lag(s): 0-3	NO_2 24-h avg (ppb): Mean (SD) 26.48 (8.02) 5th: 15.10 25th: 20.61 Median: 25.70 75th: 31.30 95th: 4102 # Stations: 2	PM_{10} (0.64) CO (0.70) O_3 (−0.04) SO_2 (0.52)	Results reported for the percent increase in admissions for an increment of NO_2 equivalent to one IQR (11 ppb) CHF, single-pollutant model 4.22 (2.61, 5.85), lag 0 CHF, two-pollutant model 4.05 (1.83, 6.31), adjusted for PM_{10} −0.37 (−2.59, 1.89), adjusted for CO 3.73 (2.10, 5.39), adjusted for O_3 3.79 (1.93, 5.67), adjusted for SO_2 CHF admission was 3 x higher among those with history of MI.

TABLE AX6.4-1 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: UNITED STATES AND CANADA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Zanobetti and Schwartz (2006) Boston, MA 1995-1999	Outcome(s) (ICD9): MI 410. Admissions through the emergency room from Medicare claims. Age group analyzed: 65+ yrs Study Design: Case crossover, control days matched yr, mo and temperature Statistical Analysis: Conditional logistic regression N: 15,578 Covariates: temperature (regression spline), day of wk Seasons: Hot (April-September) and cold Software: SAS Lags: 0, 0-1 previous day avg	NO_2 24-h avg ppb 5th: 12.59 25th: 18.30 Median: 23.20 75th: 29.13 95th: 90th-10th: 20.41 # Stations: 4	O_3 (−0.14) BC (0.70) CO (0.67) $PM_{2.5}$ (0.55) PM non-traffic (0.14) (residuals from model of $PM_{2.5}$ regressed on BC)	Results reported for percent increase in admissions for incremental increase in NO_2 equivalent to the 90th-10th percentiles (20.41 or 16.80 for 0-1, previous day avg). MI 10.21 (3.82, 15.61), lag 0 12.67 (5.82, 18.04), lag 0-1, previous day avg Results suggest traffic exposure is responsible for the observed effect. Effects more pronounced in the summer season.

* Default GAM

AMI Acute Myocardial Infarction
ARR Arrhythmia
BC Black Carbon
COH coefficient of haze
CP Course Particulate
CVD Cardiovascular Disease
EC Elemental Carbon
FP Fine Particulate
HS Hemorrhagic Stroke
ICD9 International Classification of Disease, 9th Revision
IHD Ischemic Heart Disease
IS ischemic stroke
MI Myocardial Infarction
OC Organic Carbon
OHC Oxygenated Hydrocarbons
PERI Peripheral Vascular and Cerebrovascular Disease
PM Particulate Matter
PIH primary intracerebral hemorrhage
PNC Particle Number Concentration
SHS Subarachnoid hemorrhagic stroke
TP Total Particulate
UBRE Unbiased Risk Estimator

TABLE AX6.4-2. HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: AUSTRALIA AND NEW ZEALAND

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Barnett et al. (2006) Australia and New Zealand: Brisbane, Canberra, Melbourne, Perth, Sydney Period of Study: 1998-2001	Outcome(s) (ICD9): All CVD 390-459; ARR 427; Cardiac disease 390-429; Cardiac failure 428; IHD 410-413; MI 410; Stroke 430-438. Ages groups analyzed: 15-64 yrs, ≥65yrs Study Design: Time stratified, case-crossover, multicity study # of Hospitals: All ER admissions from state government health departments Statistical Analyses: Random effects meta analysis, heterogeneity assessed using I^2 statistic. Covariates: Matched analysis controlling for long-term trend, seasonal variation and respiratory epidemics. Temperature (current-previous day) and relative humidity, pressure, extremes of hot and cold, days of wk, holidays, day after holiday, rainfall in some models. Matched on copollutants. Statistical Package: SAS Lag: 0-3	NO_2 (ppb) 1-h avg: 15.7, 23-2 24-h avg: 7.1, 11.5 IQR: 5.1 # of Stations: 1-13 depending on the city	PM_{10} 24 h CO 24 h SO_2 24 h O_3 8 h BS 24 h Matched analysis conducted to control for copollutants	Results reported for % change in hospital admissions associated with one IQR increase in NO_2 Arrhythmia ≥65: 0.4 (−1.8, 2.6) 15-64: 5.1 (2.2, 8.1) Cardiac ≥65: 3.4 (1.9, 4.9) 15-64: 2.2 (0.9, 3.4) Cardiac failure ≥65: 6.9 (2.2, 11.8) 15-64: 4.6 (0.1, 6.1) IHD ≥65: 2.5 (1.0, 4.1) 15-64: 0.7 (−1.0, 2.4) MI ≥65: 4.4 (1.0, 8.0) 15-64: 1.7 (−1.1, 2.4) All CVD ≥65: 3.0 (2.1, 3.9) 15-64: 1.7 (0.6, 2.8) NO_2 association became smaller when matched with CO. Authors hypothesize that NO_2 is a good surrogate for PM which may explain these associations.

TABLE AX6.4-2 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: AUSTRALIA AND NEW ZEALAND

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Simpson et al. (2005b) Australia (Brisbane, Melbourne, Perth, Sydney). Study Period: Jan 1996-Dec 1999	Outcome(s) (ICD9): Cardiac disease 390-429; IHD 410-413; stroke 430-438. Study Design: Time series. Statistical analysis: APHEA2 protocol, GAM (did not indicate use of stringent convergence criteria), GLM with natural splines, penalized splines. Random effects meta-analysis with tests for homogeneity. Age groups analyzed: All, 15-64, 65+ Covariates: long-term trend, temperature, humidity, day of wk, holidays, influenza epidemics Software package: SPLUS, R Lag(s): 1-3 days	NO 1-h max (ppb): Mean (range): Brisbane: 21.4 (2.1, 63.3) Sydney: 23.7 (6.5, 59.4) Melbourne: 23.7 (4.4, 66.7) Perth: 16.3 (1.9, 41.0)	PM_{10} 24 h $PM_{2.5}$ BS 24 h (0.29, 0.62) O_3 1 h CO 8 h Not all correlations reported. NO_2 affect attenuated slightly when modeled with BS but not with O_3 May be confounding of NO_2 effect by particulate.	Single-city results reported for percent increase for an increment in 1-h max NO_2 equivalent to one IQR. Pooled results reported for an increment of 1 ppb NO_2. Cardiac All ages: 1.0023 (1.0016, 1.0030), lag 0-1 15-64: 1.0015 (1.0006, 1.0025), lag 0 ≥65: 1.0018 (1.0011, 1.0025), lag 0-1 IHD All ages: 1.0019 (1.0010, 1.0027) ≥65: 1.0017 (1.0007, 1.0027) No effect for stroke. Heterogeneity in CVD results among cities, probably due to different pollutant mixtures, may have affected the results.
Hinwood et al. (2006) Perth, Australia Study Period: 1992-1998	Outcome(s): All CVD unscheduled admissions. Obtained from discharge records using ICD9 Codes. Age groups analyzed: all ages, 65+ Study design: Case crossover, time stratified with 3-4 controls within same mo Statistical Analysis: conditional logistic regression N: 26.5 daily CVD admissions Seasons: Nov-April, May-Oct	NO_2 24 h (ppb) Mean: 10.3 SD: 5 10th percentile: 4.4 90th percentile: 17.1 NO_2 1-h max (ppb) Mean: 24.8 SD: 10.1 10th percentile: 13.3 90th percentile: 37.5 # of Stations: 3	O_3 1 h, 8 h (−.06) CO 8 h (.57) BSP 24 h (.39)	Results reported for OR per incremental increase of 1 ppb NO_2. All CVD (estimated from graph) NO_2 24 h ≥65: 1.005 (1.001, 1.006), lag 1 NO_2 8 h All ages: 1.0045 (1.0012, 1.0075), lag 1 NO_2 8 h ≥65: 1.0036 (1.001, 1.0065), lag 1

TABLE AX6.4-2 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: AUSTRALIA AND NEW ZEALAND

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Jalaludin et al. (2006) Sydney, Australia Period of Study: Jan 1997-Dec 2001	Outcome(s) (ICD9): All CVD 390-459; cardiac disease 390-429; IHD 410-413; and cerebrovascular disease or stroke 430-438; Emergency room attendances obtained from health department data. Age groups included: 65+ Study Design: Time series, multi-city APHEA2 Protocol. Statistical Analysis: GAM (with appropriate convergence criteria) and GLM Models. Only GLM presented. Lag: 0-3 Covariates: daily avg temperature and daily relative, humidity, long-term trends, seasonality, weather, day of wk, public school holidays, outliers and influenza epidemics. Dose response: quartile analysis Season: Separate analyses for warm (November-April) and cool periods (May-October).	NO_2 1-h avg Mean: 32.2 SD: 7.4 Min: 5.2 Q1: 18.2 Median: 23 Q3: 27.5 Max: 59.4 # of Stations: 14	BS 24-h avg (0.35) PM_{10} 24-h avg (0.44) $PM_{2.5}$ 24-h avg (0.45) CO 8-h avg (0.55) O_3 1-h avg (0.45) SO_2 24-h avg (0.56) Two-pollutant models to adjust for copollutants	Results reported for % change in hospital admissions associated with one IQR increase in 24 h NO_2. All CVD 2.32 (1.45, 3.19), lag 0 Cardiac Disease 2.00 (0.81, 3.20), lag 0 IHD 2.11 (0.34, 3.91), lag 0 Stroke −1.66 (−3.80, 0.51) lag 0 Effect of NO_2 attenuated when CO was included in the model. NO_2 effect most prominent during the cool season.

TABLE AX6.4-2 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: AUSTRALIA AND NEW ZEALAND

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Morgan et al. (1998a) Sydney, Australia Study Period: Jan 1990-Dec 1994	Outcome(s) (ICD9): Heart Disease 410, 413, 427, 428. Inpatient statistics database for New South Wales Health Department. Study Design: Time series Statistical Analysis: Poisson regression, GEE # Hospitals: 27 Covariates: daily mean temperature, dew point temperature Lag(s): 0-2 days, cumulative Statistical Software: SAS	NO_2 24-h avg (ppb): Mean (SD): 15 (6) IQR: 11 ppb 10th-90th: 17 NO_2 1-h max (ppb): Mean (SD): 29 (3) 10-90th: 29 ppb NO_2 24-h max: 52 NO_2 1-h max: 139 # Stations: 3-14 (1990-1994)	O_3 1-h max (−0.086) PM (0.533, 0.506) Correlations for 24-h avg NO_2 concentrations Multipollutant models	Results reported as percent increase in admissions associated with an incremental increase in 1-h max NO_2 equivalent to the 10th-90th percentile. Heart Disease: 6.71 (4.25, 9.23), single pollutant, lag 0, 1-h max 6.68 (3.61, 9.84), single pollutant, lag 0, 1-h max Results lost precision but did not change substantially when stratified by age or when 24-h averaging time was used.

TABLE AX6.4-2 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: AUSTRALIA AND NEW ZEALAND

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Petroeschevsky et al. (2001) Brisbane, Australia Study Period: Jan 1987-Dec 1994, 2,922 days	Outcome(s) (ICD9): CVD 390-459. Hospital admissions, non-residents excluded. Study Design: Time series Statistical Analyses: Poisson regression, APHEA protocol, linear regression and GEEs Age groups analyzed: 15-64, 65+ Covariates: temperature, humidity, rainfall. Long-term trends, season, flu, day of wk, holidays. Statistical Software: SAS Lag(s): lag 0-4, 3-day avg, 5-day avg	NO_2 1-h max (pphm) Summer Mean: 206 Min: 0.35 Max: 5.8 Fall Mean: 2.56 Min: 0.70 Max: 5.85 Winter Mean: 3.54 Min: 0.35 Max: 8.05 Spring Mean: 3.12 Min: 0.55 Max: 15.58 Overall Mean: 2.82 Min: 0.35 Max: 15.58	BSP O_3 SO_2 Correlation between pollutants not reported.	Results reported for RR for CVD emergency admissions associated with a one-unit increase in NO_2 1-h max. CVD 15-64 yrs 0.986 (0.968, 1.005), lag 3 CVD 65+ yrs 0.990 (0.977, 1.003) CVD all ages 0.987 (0.976, 0.998)

* Default GAM

AMI Acute Myocardial Infarction
ARR Arrhythmia
BC Black Carbon
COH coefficient of haze
CP Course Particulate
CVD Cardiovascular Disease
EC Elemental Carbon
FP Fine Particulate
HS Hemorrhagic Stroke
ICD9 International Classification of Disease, 9th Revision
IHD Ischemic Heart Disease
IS ischemic stroke
MI Myocardial Infarction
OC Organic Carbon
OHC Oxygenated Hydrocarbons
PERI Peripheral Vascular and Cerebrovascular Disease
PM Particulate Matter
PIH primary intracerebral hemorrhage
PNC Particle Number Concentration
SHS Subarachnoid hemorrhagic stroke
TP Total Particulate
UBRE Unbiased Risk Estimator

TABLE AX6.4-3. HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Ballester et al. (2006) Multi-city, Spain: Barcelona, Bilbao, Castellon, Gijon, Huelva, Madrid, Granada, Oviedo, Seville, Valencia, Zaragoza Period of Study: 1995/1996-1999, N = 1,096 day	Outcome(s) (ICD9): All CVD 390-459; Heart diseases 410-414,427,428. Emergency admission from hospital records. Discharge data used. Study Design: Time series, meta-analysis to pool cities N: daily mean admissions reported by city Statistical Analyses: Poisson regression and GAM, with stringent convergence criteria, meta-analysis with fixed effect model. Tested linearity by modeling pollutant in linear and non-linear way (spline smoothing). Linear model provided best results 55% of time but used in all cases to facilitate comparability. Covariates: temperature, humidity and influenza, day of wk unusual events, seasonal variation and trend of the series Seasons: Hot: May to October; Cold: November to April Statistical Package: SPLUS Lag: 0-3	NO_2 24-h avg ($\mu g/m^2$): Mean: 51.5 10th percentile: 29.5 90th percentile: 74.4 # of Stations: Depends on the city Correlation among stations: NR	CO 8-h max (0.58) O_3 8-h max (−0.03) SO_2 24 h (0.46) BS 24 h (0.48) TSP 24 h (0.48) PM_{10} 24 h (0.40) Two-pollutant models used to adjust for copollutants	Results reported for % change in hospital admissions associated with 10 $\mu g/m^2$ increase in NO_2 All CVD 0.38% (0.07%, 0.69%), lag 0-1 Heart Diseases: 0.86% (0.44%, 1.28%) Effect of NO_2 was diminished in two-pollutant models.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Lanki et al. (2006) Europe (Augsburg, Helsinki, Rome, Stockholm) Study period: 1992-2000	Outcome(s) (ICD9): AMI 410. Ascertained from discharge records or AMI registry data depending on the city. Study Design: Time series Statistical Analysis: Poisson regression, for non-linear confounders – penalized splines in GAM chosen to minimize UBRE score. Random-effects model for pooled estimates. N: 26,854 hospitalizations Statistical Software: R package Covariates: barometric pressure, temperature, humidity. Lag(s): 0-3 day	NO_2 ($\mu g/m^3$) Augsburg: 25th: 40.2 50th: 49.2 75th: 58.9 98th: 88.7 Barcelona 25th: 34.8 50th: 45.0 75th: 60.0 98th: 86.0 Helsinki 25th: 21.8 50th: 28.7 75th: 37.6 98th: 64.7 Rome 25th: 61.9 50th: 70.6 75th: 80.4 98th: 102.5 Stockholm 25th: 16.3 50th: 22.2 75th: 28.6 98th: 45.9	PM_{10} (0.29, 0.64) CO (0.43, 0.75) O_3 (0.17, 0.38) Range in correlations depends on the city Two-pollutant models for PNC with O_3 and PM_{10} only.	Results reported as RR associated with an incremental increase in NO_2 equivalent to the IQR (8 $\mu g/m^2$) Pooled results for 5 Cities: 0.996 (0.998, 1.015), lag 0 No significant results observed for analyses stratified by age or season for lag 0/1.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Von Klot et al. (2005) Europe (Augsburg, Barcelona, Helsinki, Rome, Stockholm) Study Period: 1992-2000	Outcome(s) (ICD9): Re-admission for AMI 410; angina pectoris 411 and 413; Cardiac diseases including AMI angina pectoris, dysrhythmia (427), heart failure (428). Hospital admissions database used to identify cases. Population: Incident cases of MI during 1992-2000 among those ≥35 yrs old. N Augsburg: 1560 N Barcelona: 1134 N Helsinki: 4026 N Rome: 7384 N Stockholm: 7902 Study Design: Prospective Cohort Statistical Analyses: Poisson regression, at risk period from the 29th day after the index event until the event of interest, death, migration or loss to follow-up. GLM models, penalized spline functions for continuous confounders. City results pooled using random-effects model. Heterogeneity assessed. Sensitivity analyses conducted varying the smooth functions, convergence criteria, and how confounders were specified. Statistical Software: R package Covariates: daily mean temperature, dew point temperature, barometric pressure, relative humidity, vacations or holidays. Lag: 0-3 days	NO_2 24-h avg ($\mu g/m^2$): Augsburg Mean: 49.6 5th: 30 25th: 39.7 75th: 57.2 95th: 75.3 Barcelona Mean: 47.7 5th: 18 25th: 34.0 75th: 60 95th: 83 Helsinki Mean: 30.1 5th: 13 25th: 21.2 75th: 36.7 95th: 52.9 Rome Mean: 15.8 5th: 5.4 25th: 10.1 75th: 21.7 95th: 25.9 Stockholm Mean: 22.8 5th: 10.3 25th: 16 75th: 28 95th: 39.4 # Stations: 1-5	CO 24 h (0.44, 0.75) O_3 8 h (−0.2, −0.13) PM_{10} (.29, .66) PNC (.44, .83) Two-pollutant models but NO_2, CO, and PNC not modeled together because they were too highly correlated.	Results reported for RR for incremental increases in same day NO_2 equivalent to the mean of the city specific IQR's multiplied by 0.05 (8 $\mu g/m^2$). Pooled results are below: MI 1.028 (0.997, 1.060) Angina Pectoris 1.032 (1.006, 1.058) Cardiac Diseases 1.032 (1.014, 1.051) Two-pollutant models show that the effect of NO_2 independent of PM_{10} and O_3. Traffic exhaust may be associated with cardiac readmission.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Atkinson et al. (1999a) London, England Period of Study: 1992-1994, N = 1,096 day	Outcome(s) (ICD9): All CVD 390-459; IHD 410-414. Emergency admissions obtained from the Hospital Episode Statistics (HES) database. Ages groups analyzed: 0-14 yr, 15-64 yr, 0-64 yr, 65+ yr, 65-74 yr, 75+ yr Study Design: Time series, hospital admission counts N: 189,109 CVD admissions Catchment area: 7 million residing in 1600 Km^2 area of Thames basin. Statistical Analyses: APHEA protocol, Poisson regression Covariates: adjusted long-term seasonal patterns, day of wk, influenza, temperature, humidity (compared alternative methods for modeling meteorological including linear, quadradic, piece-wise, spline) Seasons: warm season April-September, cool season remaining mos, interactions between season investigated Dose response investigated: yes, bubble charts presented Statistical Package: SAS Lag: 0-3	1-h max (ppb) Mean: 50.3 SD: 17.0 Min: 22.0 Max: 224.3 10th-90th percentile: 36 # of Stations: 3, results averaged across stations Correlation among stations: 0.7-0.96	PM_{10} 24 h CO 24 h SO_2 24 h O_3 8 h BS 24 h Correlations of NO_2 with CO, SO_2, O_3, BS ranged from 0.6-0.7 Correlation of NO_2 with O_3 negative Two-pollutant models used adjust for copollutants	Results reported for % change in hospital admissions associated with 10th-90th percentile increase in NO_2 (36 ppb) All CVD Ages 0-64: 1.20% (−0.62%, 3.05%), lag 0 Ages 65+: 1.68% (0.32%, 3.06%), lag 0 IHD Ages 0-64: 1.53% (−1.22%, 4.37%), lag 0 Ages 65+: 3.03% (0.87%, 5.24), lag 0 NO_2 was associated with increased CVD admissions for all ages but this association was stronger among those 65+ yrs old. Similar increase associated with IHD among those 65+ yrs old. Monitors close to roadways were not used in the study. Correlations for NO_2 between urban monitoring sites were high. Authors suggest that the pollution levels are uniform across the study area. Authors did not investigate the interaction between meteorological variables and air pollution. In two pollutant models, O_3 had little impact on NO_2. BS moderated the association of NO_2 with CVD among the 65+ age group. Suggestion that NO_2 associations were non-linear.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Ballester et al. (2001) * Valencia, Spain Period of Study: 1992-1996	Outcome(s) (ICD9): All CVD 390-459; heart diseases 390-459; cerebrovascular diseases 430-438. Admissions from city registry – discharge codes used. Study Design: Time series N: 1080 CVD admissions # of Hospitals: 2 Catchment area: 376,681 inhabitants of Urban Valencia Statistical Analyses: Poisson regression, GAM, APHEA/ Spanish EMECAM protocol. Both Linear and non parametric model, including a loess term was fitted, departure from linearity assess by comparing deviance of both models. Covariates: long-term trend and seasonality, temperature and humidity, wk days, flu, special events, air pollution. Seasons: Hot season May to Oct.; Cold season Nov to April Statistical Package: SAS Lag: 0-4	1-h max ($\mu g/m^2$) Mean: 116.1 SD: NR Min: 21.1 Max: 469.0 median: 113.2 # of Stations: 14 manual, 5 automatic Correlation among stations: 0.3-0.62 for BS, 0.46-0.78 for gaseous pollutants	CO 24 h (0.03) SO_2 24 h (0.33) O_3 8 h (−0.26) BS (0.33) Two-pollutant models used to adjust for copollutants	Results reported for RR corresponding to a 10 $\mu g/m^2$ increase in NO_2 All CVD 1.0302 (1.0042, 1.0568), lag 0 Heart Disease 1.0085 (0.9984, 1.0188), lag 2 Cerebrovascular Disease 1.0362 (1.0066, 1.0667), lag 4 Clear association of NO_2 with cerebrovascular disease observed. Association persisted after Inclusion of BS and SO_2 in two-pollutant models with NO_2. Cases of digestive disorders served as a control group - null association with NO_2 observed.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
D'Ippoliti et al. (2003) Rome, Italy Study Period: Jan 1995- June 1997	Outcome(s) (ICD): AMI 410 (first episode). Computerized hospital admission data. Study Design: Case crossover, time stratified, control days within same mo falling on the same day. Statistical Analyses: Conditional logistic regression, examined homogeneity across co-morbidity categories N: 6531 cases Age groups analyzed: 18-64 yrs, 65-74 yrs, ≥75 Season: Cool: October-March; Warm: April-September. Lag(s): 0-4 day, 0-2 day cum avg Dose Response: OR for increasing quartiles presented and p-value for trend.	NO_2 24 h ($\mu g/m^3$) Mean (SD): 86.4 (15.8) 25th: 74.9 50th: 86.0 75th: 96.9 IQR: 22 # Stations: 5	TSP 24 h (0.37) SO_2 24 h (0.31) CO 24 h (0.03) No multipollutant models	Results presented for OR associated with incremental increase in NO_2 equivalent to one IQR. AMI 1.026 (1.002, 1.052), lag 0 1.026 (0.997, 1.057), lag 0-2 Association observed for NO_2 but TSP association more consistent. Authors think that TSP, CO, and NO_2 cannot be distinguished from traffic-related pollution in general.
Llorca et al. (2005) Torrelavega, Spain Study period: 1992-1995	Outcome(s) (ICD): CVD (called cardiac in paper) 390-459. Emergency admissions, excluding non residents. Obtained admissions records from hospital admin office. Study design: Time series Statistical analyses: Poisson regression, APHEA protocol Covariates: rainfall, temperature, wind speed direction N: 18,137 admissions Statistical software: STATA Lag(s): not reported	NO_2 24 h $\mu g/m^3$: Mean (SD): 21.3 (16.5)	TSP (−0.12) SO_2 (0.588) SH_2 (0.545) NO (0.855) Multipollutant models	Results reported for RR of hospital admissions for 100 $\mu g/m^3$ increase in NO_2. Cardiac admissions: 1.27 (1.14, 1.42), 1-pollutant model 1.10 (0.92, 1.32), 5-pollutant model Effect of NO_2 diminished in multipollutant model.

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Pantazopoulou et al. (1995) Athens, Greece Study Period: 1988 (Winter and Summer)	Outcome(s): Cardiac Disease ICD codes not provided. Cases ascertained from National Center for Emergency Service database. Cases diagnosed at time of admission so they are ED visits and were not necessarily admitted to the hospital. Study design: Time series Statistical Analyses: Linear regression (not well described) Covariates: Dummy variables for winter mos with January as referent. Dummy variables for summer mos with April as referent. Day of the wk, holidays, temperature, relative humidity, N: 25,027 cardiac admissions. Lag(s): NR	NO_2 1-h max ($\mu g/m^3$): Winter Mean (SD): 94 (25) 5th: 59 50th: 93 95th: 135 Summer Mean (SD): 111 (32) 5th: 65 50th: 108 95th: 173 # Stations: 2	CO, BS No correlations provided	Results reported for regression coefficients based on an incremental increase in NO_2 of 76 $\mu g/m^3$ in winter and 108 $\mu g/m^3$ in summer (5th to 95th percentile). Winter (regression coefficient) 11.2 (3.3, 19.2) Summer (regression coefficient) −0.06 (−6.6, 6.5)
Poloniecki et al. (1997) London, UK Study Period: April 1987-March 1994, 7 yrs	Outcome(s): All CVD 390-459; MI 410; Angina pectoris 413; other IHD 414; ARR 427; congestive heart failure 428; cerebrovascular disease 430-438. Hospital Episode Statistics (HES) data on emergency hospital admissions. Study Design: Time series N: 373, 556 CVD admissions Statistical Analyses: Poisson regression with GAM, APHEA protocol Covariates: long term trends, seasonal variation, day of wk, influenza, temperature and humidity. Season: Warm, April-September; Cool, October-March. Lag: 0-1 day	NO_2 24 h ppb: Min: 8 10%: 23 Median: 35 90%: 53 Max: 198	Black Smoke CO 24 h SO_2 24 h O_3 8 h Correlations between pollutants high but not specified.	Results expressed as a relative rate (RR) for an incremental increase of NO_2 equivalent to 30 ppb (10th-90th percentile) AMI: 1.0274 (1.0084, 1.0479) Angina Pectoris: 1.0212 (0.9950, 1.0457) Other IHD: 0.99 (0.0067, 1.0289) Cardiac ARR: 1.0274 (1.0006, 1.0984) Heart Failure: 0.9970 (0.9769, 1.0194) Cerebrovascular Disease: 0.9851 (0.9684, 1.0045) Other Circulatory: 1.0182 (1.0000, 1.0398) All CVD: 1.0243 (1.0054, 1.0448) No attenuation of NO_2 association with MI in two-pollutant model (cool season).

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Pönkä and Virtanen (1996) Helsinki, Finland Study Period: 1987-1989, 3 yrs	Outcome(s) (ICD9): IHD 410-414; MI 410; TIA 411; Cerebrovascular diseases 430-438; Cerebral ischemia due to occlusion of extracerebral vessels 433; Cerebral ischemia due to occlusion of cerebral vessels 434; Transient ischemic cerebral attack 435. Case ascertainment was for both emergency admission and hospital admissions – done via registry system. Study Design: Time series Statistical Analyses: Poisson regression, pollutant concentrations log transformed N: 12,664 all IHD admissions; 7005 IHD ED admissions; 7232 cerebrovascular hospital admissions; 3737 cerebrovascular ED admissions. Covariates: weather, day of wk, long-term trends, influenza Lag(s): 1-7 days	NO_2 8 h (µg/m^3) Mean (SD): 39 (16.2) Range: 4, 170 NO 8 h µg/m^3 Mean (SD): 91 (61) Range: 7, 467 # Stations: 2	SO_2 8 h NO 8 h TSP 8 h O_3 8 h NO_2 highly correlated with SO_2 and TSP	Results reported are regression coefficients and standard errors (SE). NO_2 with ED admissions for transient short term ischemic attack −0.056 (0.105), p = 0.59, lag 1 NO_2 with ED admissions for cerebrovascular disease −0.025 (0.057), p = 0.657, lag 1 NO with IHD, all admissions 0.097 0.023, p < 0.001, lag 1 NO with IHD, ED admissions 0.111 (0.030), p < 0.001, lag 1 Significant increase in admissions for transient short-term ischemic attack and cerebrovascular diseases for lag 6 associated with NO_2 exposure.
Prescott et al. (1998) * Edinburgh, UK Study period: Oct 1992-June 1995	Outcome(s) (ICD9): Cardiac and cerebral ischemia 410-414, 426-429, 434-440. Extracted from Scottish record linkage system. Study Design: Time series Statistical Analysis: Poisson, log linear regression models Age groups analyzed: <65, 65+yrs Covariates: seasonal and wkday variation, temperature, and wind speed. Lag(s): 0, 1, 3 day moving avg	NO_2 24 h (ppb) Mean (SD): 26.4 (7.0) Range: 9, 58 IQR: 10 ppb	O_3, 24 h PM, 24 h SO_2, 24 h CO, 24 h Correlations not reported.	Results reported for percent change in admissions based on an incremental increase in NO_2 equivalent to the IQR of 10 ppb. <65 yrs, CVD admissions −0.05 (−5.2, 4.5), 3 day moving avg 65+ yrs, CVD admissions −0.9 (−8.2, 7.0), 3 day moving avg Data for lag 1 not presented

TABLE AX6.4-3 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: EUROPE

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Yallop et al. (2007) London, England Study Period: Jan. 1988-Oct. 2001, >1400 days	Outcome(s): Acute pain in Sickle Cell Disease (HbSS, HbSC, HbS/β0, thalassaemia, HbS/β+). Admitted to hospital for at least one night. Study Design: Time series Statistical Analyses: Cross-correlation function N: 1047 admissions Covariates: no adjustment made in analysis, discussion includes statement that the effects of weather variables and copollutants are inter-related. Statistical Package: SPSS Lag(s): 0-2 days Dose response: quartile analysis, graphs presented, ANOVA comparing means across quartiles.	NR	O_3, CO, NO, NO_2, PM_{10}: daily avg used for all copollutants High O_3 levels correlate with low NO, low CO, increased wind speeds and low humidity and each was associated with admission for pain. Not possible to distinguish associations in analysis.	Results reported are cross-correlation coefficients. NO inversely correlated with admission for acute pain in SCD. CFF: −0.063, lag 0

* Default GAM

AMI Acute Myocardial Infarction
ARR Arrhythmia
BC Black Carbon
COH coefficient of haze
CP Course Particulate
CVD Cardiovascular Disease
EC Elemental Carbon
FP Fine Particulate
HS Hemorrhagic Stroke
ICD9 International Classification of Disease, 9th Revision
IHD Ischemic Heart Disease
IS ischemic stroke
MI Myocardial Infarction
OC Organic Carbon
OHC Oxygenated Hydrocarbons
PERI Peripheral Vascular and Cerebrovascular Disease
PM Particulate Matter
PIH primary intracerebral hemorrhage
PNC Particle Number Concentration
SHS Subarachnoid hemorrhagic stroke
TP Total Particulate
UBRE Unbiased Risk Estimator

TABLE AX6.4-4. HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Chan et al. (2006) * Taipai, Taiwan Period of Study: April 1997-Dec 2002, 2090 days	Outcome(s) (ICD9): Cerebrovascular disease 430-437; stroke 430-434; hemorrhagic stroke 430-432; ischemic stroke 433-434. Emergency admission data collected from National Taiwan University Hospital. Ages groups analyzed: age >50 included in study Study Design: Time series N: 7341 Cerebrovascular admissions among those >50 yrs old # of Hospitals: Catchment area: Statistical Analyses: Poisson regression, GAMs used to adjust for non-linear relation between confounders and ER admissions. Covariates: time trend variables: yr, mo, and day of wk, daily temperature difference, and dew point temperature. Linearity: Investigated graphically by using the LOESS smoother. Statistical Package: NR Lag: 0-3, cumulative lag up to 3 days	NO_2 24-h avg (ppb): Mean: 29.9 SD: 8.4 Min: 8.3 Max: 77.1 IQR: 9.6 ppb # of Stations: 16 Correlation among stations: NR	PM_{10} 24 h, r = 0.50 $PM_{2.5}$ 24 h, r = 0.64 CO 8-h avg, r = 0.77 SO_2 24 h, r = 0.64 O_3 1-h max, r = 0.43 Two-pollutant models to adjust for copollutants	Results reported for OR for association of emergency department admissions with an IQR increase in NO_2 (9.3 ppb) Cerebrovascular: 1.032 (0.991, 1.074), lag 0 Stroke: 0.994 (0.914, 1.074), lag 0 Ischemic stroke: 1.025 (0.956, 1.094), lag 0 Hemorrhagic stroke: 0.963 (0.884, 1.042), lag 0 No significant associations for NO_2 reported. Lag 0 shown but similar null results were obtained for lags 1-3. NO_2 highly correlated with PM and CO.

TABLE AX6.4-4 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Chang et al. (2005) Taipei, Taiwan Study Period: 1997-2001, 5 yrs	Outcome(s) (ICD9): CVD 410-429. Daily clinic visits or hospital admission from computerized records of National Health Insurance. Discharge data. Source Population: 2.64 Million N: 40.8 admissions/day, 74,509/5 yrs # Hospitals: 41 Study Design: case crossover, referent day 1 wk before or after index day Statistical Analyses: conditional logistic regression. Covariates: same day temperature and humidity. Season: warm/cool (stratified by temperature cutpoint of 20 ºC) Lag(s): 0-2 days	NO_2 24-h avg (ppb): Mean: 31.54 Min: 8.13 25th: 26.27 50th: 31.03 75th: 36.22 Max: 77.97 # of Stations: 6	CO 24-h avg O_3 24-h avg SO_2 24-h avg PM_{10} 24-h avg Correlations not reported. Two-pollutant models to adjust for copollutants	OR for the association of CVD admissions with an incremental increase in NO_2 equivalent to one IQR. Warm (≥20 ºC) 1.177 (1.150, 1.205) Cool (<20 ºC) 1.112 (1.058, 1.168) NO_2 effect remained in all warm season two-pollutant models. Effect remained in cool season two-pollutant models with the exception of the model that included PM_{10}.

TABLE AX6.4-4 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Lee et al. (2003a) Seoul, Korea Study period: Dec 1997-Dec 1999, 822 days, 184 days in summer	Outcome(s) (ICD10): IHD: Angina pectoris 120; Acute or subsequent MI 121-123; other acute IHD 124. Electronic medical insurance data used. Study Design: Time series Statistical Methods: Poisson regression, GAM with strict convergence criteria. Age groups analyzed: all ages, 64+ Covariates: long-term trends LOESS smooth, temperature, humidity, day of wk. Season: Presented results for summer (June, July, August) and entire period. Lag(s): 0-6	NO_2 24 h (ppb): 5th: 16 10th: 23.7 Median: 30.7 75th: 38.3 95th: 48.6 Mean (SD): 31.5 (10.3) IQR: 14.6	PM_{10}, r = 0.73, 0.74 SO_2, r = 0.72, 0.79 O_3, r = –0.07, 0.63 CO, r = 0.67, 0.79 Range depends on summer vs. entire period. Two-pollutant models	Results reported for RR of IHD hospital admission for an incremental increase in NO_2 equivalent to one IQR. 64+, entire study period: 1.08 (1.03, 1.14), lag 5 64+, summer only: 1.25 (1.11, 1.41), lag 5 Results for lag 5 presented above. Lag 0 or 1 results largely null – presented graphically. Confounding by PM_{10} was not observed in these data using two-pollutant models.
Tsai et al. (2003a) Kaohsiung, Taiwan Study period: 1997-2000	Outcome(s) (ICD9): All cerebrovascular 430-438; SHS 430; PIH 431-432; IS 433-435; Other 436-438. Ascertained from National Health Insurance Program computerized admissions records. Study Design: Case crossover Statistical Analysis: Conditional logistic regression. Statistical Software: SAS Seasons: ≥20 ºC; <20 ºC. N: 23,179 stroke admissions # Hospitals: 63 Lag(s): 0-2, cumulative lag up to 2 previous days	NO_2 (ppb) Min: 6.25 25th: 19.25 Median: 28.67 75th: 36.33 Max: 63.40 Mean: 28.17	PM_{10} SO_2 CO O_3	Results reported as OR for the association of admissions with an incremental increase of NO_2 equivalent to the IQR of 17.1 ppb PIH admissions Warm: 1.56 (1.32, 1.84), lag 0-2 Cool: 0.81 (0.0, 1.31), lag 0-2 IS admissions: Warm: 1.55 (1.40, 1.71), lag 0-2 Cool: 1.16 (0.81, 1.68), lag 0-2 Effects persisted after adjustment for PM_{10}, SO_2, CO, and O_3.

TABLE AX6.4-4 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Wong et al. (1999) Hong Kong, China Study Period: 1994-1995	Outcome(s) (ICD9): CVD: 410-417, 420-438, 440-444; CHF 428; IHD 410-414; Cerebrovascular Disease 430-438. Hospital admissions through ER departments via Hospital Authority (discharge data). Study Design: Time series Statistical Analyses: Poisson regression, APHEA protocol # Hospitals: 12 Covariates: daily temperature, relative humidity day of wk, holidays, influenza, long-term trends (yr and seasonality variables). Interaction of pollutants with cold season examined. Season: Cold (Dec-March) Lag(s): 0-3 days	NO_2 24-h avg ($\mu g/m^3$) Mean: 51.39	PM_{10}, r = 0.79 SO_2 O_3 Range for other pollutants: r = 0.68, 0.89. Two-pollutant models	Results reported for RR associated with incremental increase in NO_2 equal to 10 $\mu g/m^3$. CVD 65+ yrs: 1.016 (1.009, 1.023) All ages: 1.013 (1.007, 1.020) CHF 1.044 (1.25, 1.063) IHD 1.010 (0.999, 1.020) Cerebrovascular Disease 1.008 (0.998, 1.018) Interaction of NO_2 with O_3 observed

TABLE AX6.4-4 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Yang et al. 2004a Kaohsiung, Taiwan Period of Study: 1997-2000	Outcome(s) (ICD9): All CVD: 410-429 * (All CVD typically defined to include ICD9 codes 390-459) N: 29,661 Study Design: Case crossover Statistical Analysis: Poisson time-series regression models, APHEA protocol # of Hospitals: 63 Seasons: authors indicate not considered because the Taiwanese climate is tropical with no apparent seasonal cycle Covariates: stratified by warm (≥25º) and cold (<25º) days, temperature and humidity measurements included in the model Statistical Package: SAS Lag: 0-2 days	Min: 6.25 ppb 25%: 19.25 ppb 50%: 28.67 ppb 75%: 36.33 ppb Max: 63.40 ppb Mean: 28.17 ppb # of Stations: 6 Correlation among stations: NR	PM_{10} CO SO_2 O_3 8 Two-pollutant models used to adjust for copollutants Correlations NR	OR's for the association of one IQR (17.08 ppb) increase in NO_2 with daily counts of CVD hospital admissions are reported All CVD (ICD9: 410-429), one-pollutant model ≥25º: 1.380 (1.246, 1.508) <25º: 2.215 (2.014, 2.437) All CVD (ICD9: 410-429), two-pollutant models Adjusted for PM_{10}: ≥25º: 1.380 (1.246, 1.508) <25º: 2.215 (2.014, 2.437) Adjusted for SO_2: ≥25º: 1.149 (1.017, 1.299) <25º: 2.362 (2.081, 2.682) Adjusted for CO ≥25º: 1.039 (0.919, 1.176) <25º: 2.472 (2.138, 2.858) Adjusted for O_3 ≥25º: 1.159 (1.051, 1.277) <25º: 2.243 (2.037, 2.471) Association of CVD admissions with NO_2 attenuated on warm days after adjustment for copollutants. Association persisted on cool days. Kaohsiung is the center of Taiwan's heavy industry.

TABLE AX6.4-4 (cont'd). HUMAN HEALTH EFFECTS OF OXIDES OF NITROGEN: CVD HOSPITAL ADMISSIONS AND VISITS: ASIA

Reference, Study Location, & Period	Outcomes, Design, & Methods	Mean Levels & Monitoring Stations	Copollutants (Correlations)	Effects: Relative Risk or Percent Change & Confidence Intervals ([95% Lower, Upper])
Ye et al. (2001) Tokyo, Japan Study Period: July-August, 1980-1995	Outcome(s): Angina 413; Cardiac insufficiency 428; Hypertension 401-405; MI 410. Diagnosis made by attending physician for hospital emergency transports. Age groups analyzed: 65+ yrs male and female Statistical analysis: GLM Covariates: maximum temperature, confounding by season minimal since only 2 summer mos included in analysis Statistical Software: SAS Lag(s): 1-4 days	NO_2 24-h avg (ppb) Minimum: 5.3 Maximum: 72.2 Mean (SD): 25.4 (11.4)	O_3, r = 0.183 PM_{10}, r = 0.643 SO_2, r = 0.333 CO, r = 0.759	Results reported for model coefficient and 95% CI. Angina: 0.007 (0.004, 0.009) Cardiac insufficiency: 0.006 (0.003, 0.01) MI: 0.006 (0.003, 0.01)

* Default GAM

AMI Acute Myocardial Infarction
ARR Arrhythmia
BC Black Carbon
COH coefficient of haze
CP Course Particulate
CVD Cardiovascular Disease
EC Elemental Carbon
FP Fine Particulate
HS Hemorrhagic Stroke
ICD9 International Classification of Disease, 9th Revision
IHD Ischemic Heart Disease
IS ischemic stroke
MI Myocardial Infarction
OC Organic Carbon
OHC Oxygenated Hydrocarbons
PERI Peripheral Vascular and Cerebrovascular Disease
PM Particulate Matter
PIH primary intracerebral hemorrhage
PNC Particle Number Concentration
SHS Subarachnoid hemorrhagic stroke
TP Total Particulate
UBRE Unbiased Risk Estimator

TABLE AX6.5-1. STUDIES EXAMINING EXPOSURE TO AMBIENT NO_2 AND HEART RATE VARIABILITY AS MEASURED BY STANDARD DEVIATION OF NORMAL-TO-NORMAL INTERVALS (SDNN)

Author, Year	% Change (95% CI)	Location	Subjects	Analysis Method	NO_2 Conc (ppb)			Copollutant Correlation			
					Avg Time	Mean (sd)	Range	PM	O_3	SO_2	CO
Liao et al. (2004)		US, ARIC study	4,390 adults	multivariable linear regression	24 h	21 (8)					
lag 1	−5.0% (−9.2, −.7)										
Chan et al. (2005)		Taiwan	83 adults recruited from cardiology	linear mixed effects regression	1 h	33 (15)	1, 110	PM_{10} 0.4	−0.4	0.5	0.7
4-h lag	−4.5% (−8.1, −.30)										
8-h lag	−6.9% (−12.0, −1.8)										
Wheeler et al. (2006)		Atlanta	30 adults (12 MI + 22 COPD)	linear mixed models	4 h	18 (no sd given)	p10-p20, 7, 30	$PM_{2.5}$ 0.4			0.5
MI patients [N = 12]											
4 h lag	−26.0% (−42.1, −8.6)										
COPD patients [N = 22]											
4 h lag	16.6% (0.2, 34.3)										
Luttmann-Gibson et al. (2006)		Steubenville	32 adults (>50yrs)	mixed models	24 h	10 (no sd given)	p25-p75, 6, 13	$PM_{2.5}$ 0.4	−0.3	0.3	
lag 1	0.3% (−6.0, 6.6)										
Schwartz et al. (2005)		Boston	28 elderly adults	hierarchical models	24 h	med 18	p25-p75, 14, 23	$PM_{2.5}$ 0.3	0.02		0.6
lag 1	−1.6% (−7.8, 5.1)										

All results given for 20 ppb increase in NO_2 with 24-h averaging time, or 30 ppb for 1-h averaging time. (20 ppb increases also used for averaging times between 1 and 24 h)

TABLE AX6.5-2. STUDIES EXAMINING EXPOSURE TO AMBIENT NO_2 AND HEART RATE VARIABILITY AS MEASURED BY VARIABLES RECORDED ON IMPLANTABLE CARDIOVERTER DEFIBRILLATORS (ICDS)

Author, Year	OR (95% CI)	Location	Subjects	Analysis Method	NO_2 Conc (ppb)		Copollutant Correlation			
					Mean (sd)	Range	$PM_{2.5}$	O_3	SO_2	CO
Risk of ICD discharge										
Peters et al. (2000a)		Eastern MA	100 cardiac outpatients	logistic regression, fixed effects	23 (no sd given)	11, 65	0.6	−0.3	0.3	0.7
lag 1	1.55 (1.05, 2.29)									
lag 0-4	1.88 (1.01, 3.49)									
Risk of ICD-recorded ventricular arrhythmias										
Rich et al. (2005)		Boston	203 cardiac outpatients	case-crossover	med 22	p25-max, 18, 62				
all patients										
lag 0-1	1.54 (1.11, 2.18)									
patients with recent arrhythmia (< 3 days)										
lag 0-1	2.09 (1.26, 3.51)									
Dockery et al. (2005)		Boston	307 cardiac outpatients	logistic regression, GEE	med 23	p25-p95, 19, 34	>0.4	<−0.4	>0.4	0.6
patients with recent arrhythmia (<3 days)										
lag 0-1	2.14 (1.14, 4.03)									
Risk of ST-segment depression >0.1 mV										
Pekkanen et al. (2002)		Finland	45 cardiac patients	linear regression, GAM	med 16	p25-max, 12, 36	0.4			0.3
lag 2	14.1 (3.0, 65.4)									
Risk of resting heart rate >75 bpm										
Ruidavets et al. (2005)		France	863 adults	polytomous logistic regression	16 (6)	2, 48		−0.3	0.7	
lag 8h	2.7 (1.2, 5.4)									

All results given for 20 ppb increase in NO_2 with 24-h averaging time.

TABLE AX6.6-1. BIRTH WEIGHT AND LONG-TERM NO_2 EXPOSURE STUDIES

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	
Lin et al. (2004)	Taiwan	Term LBW	92,288 birth cert													3 km
Pregnancy			1995-1997		Logistic regression		<26.1	26.1, 32.9	>32.9							
Medium NO_2				1.06 (0.93, 1.22)												
High NO_2				1.06 (0.89, 1.26)												
Trimester 1							<24.3	24.3, 34.7	>34.7							
Medium NO_2				1.10 (0.96, 1.27)												
High NO_2				1.09 (0.89, 1.32)												
Trimester 2							<24.0	24.0, 34.4	>34.4							
Medium NO_2				0.87 (0.76, 1.00)												
High NO_2				0.93 (0.77, 1.12)												
Timester 3							<23.8	23.8, 34.2	>34.2							
Medium NO_2				1.01 (0.88, 1.16)												
High NO_2				0.86 (0.71, 1.03)												
Lee et al. (2003b)	Seoul, Korea	Term LBW	388,105 birth cert		Generalized additive model (GAM)	24 h	25	31.4	39.7							
Pregnancy			1996-1998	1.04 (1.00, 1.08)												
Trimester 1				1.02 (0.99, 1.04)	Interquartile						0.66		0.75	0.77		
Trimester 2				1.03 (1.01, 1.06)							0.81		0.77	0.78		
Trimester 3				0.98 (0.96, 1.00)							0.8		0.76	0.82		
Bobak M. (2000)	Czech	LBW adjusted	69,935 birth cert		Logistic regression	24 h	12.2	20	31.1							
Trimester 1		for GA	1991 only	0.98 (0.81, 1.18)	50 µg increase								0.53			
Trimester 2				0.99 (0.80, 1.23)									0.62			
Trimester 3				0.97 (0.80, 1.18)									0.63			

TABLE AX6.6-1 (cont'd). BIRTH WEIGHT AND LONG-TERM NO_2 EXPOSURE STUDIES

Author, Year	Study Location	Study Group	Study subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants					BS	Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO		
Maroziene and Grazuleviciene (2002)	Kaunas Lithuania	LBW adjusted for GA	3,988 birth cert 1998 only		Logistic regression			6.2 (5.7)								
Pregnancy				1.28 (0.97, 1.68)	10 µg increase											
Medium NO_2				0.96 (0.47, 1.96)												
High NO_2				1.54 (0.80, 2.96)												
Trimester 1				0.91 (0.53, 1.56)	10 µg increase											
Trimester 2				0.93 (0.61, 1.41)												
Trimseter 3				1.34 (0.94, 1.92)												
Liu et al. (2003)	Vancouver	LBW adjusted	229,085 birth cert		Logistic regression	24 h	15.1	18.1	22.3			−0.25	0.61	0.72		
First mo		for GA	1986-1998	0.98 (0.90, 1.07)	10 ppb increase											
Last mo				0.94 (0.85, 1.04)												
Salam et al. (2005)	Southern CA	Term LBW	3,901 birth cert		Logistic regression			36.1 (15..4)			0.55	−0.1		0.41		5 km or 3 within 50 km
Pregnancy		CHS	1975-1987	0.8 (0.4, 1.4)				IQR 25								within county
Trimester 1				0.9 (0.5, 1.5)												
Trimester 2				1.0 (0.6, 1.6)												
Trimester 3				0.6 (0.4, 1.1)												
Bell M et al. (2007)	CT and MA	LBW adjusted	358,504 birth cert		logistic regression			17.4 (5.0)		0.64	0.55					
pregnancy		for GA	1999-2002	1.027 (1.002, 1.051)	interquartile			IQR 4.8								
black mothers				−12.7 (−18.0, −7.5)	linear regression											
white mothers				−8.3 (−10.4, −6.3)	difference in gms											
					per IQR											

TABLE AX6.6-2. PRETERM DELIVERY AND LONG-TERM NO_2 EXPOSURE STUDIES

Author, Year	Study Location	Study Group	Study subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb) Low	Mid-range	High	Correlation with Other Pollutants $PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
Bobak 2000	Czech	Preterm	69,935 birth cert		Logistic regression	24 h	12.2	20	31.1				0.62			
trimester 1			1991 only	1.10 (1.00, 1.21)	50 µg increase											
trimester 2				1.08 (0.98, 1.19)												
trimester 3				1.11 (1.00, 1.23)												
Liu S et al. (2003)	Vancouver	Preterm	229,085 birth cert			24 h	15.1	18.1	22.3			−0.25	0.61	0.72		13 monitors
first mo			1986-1998	1.01 (0.94, 1.07)	10 ppb increase											avg
last mo				1.08 (0.99, 1.17)												
Maroziene and Grazuleviciene R (2002)	Kaunas Lituania	Preterm	3,988 birth cert		Logistic regression											
pregnancy				1.25 (1.07, 1.46)	10 µg increase			6.2 (5.7)								
medium NO_2				1.14 (0.77, 1.68)												
high NO_2				1.68 (1.15, 2.46)												
trimester 1				1.67 (1.28, 2.18)	10 µg increase											
trimester 2				1.13 (0.90, 1.40)												
trimester 3				1.19 (0.96, 1.47)												
Ritz et al. (2000)	southern CA	Preterm	97,158 birth cert		Logistic regression	24 h	32	40.9	50.4		0.74	−0.12		0.64		Zipcode within 2 miles
first mo			1989-1993	No effects for												
6 wks before birth				any preg period												
Hansen C et al. (2006)	Brisbane	Preterm	28,200 birth cert		Logistic regression	24 h		8.8 (4.1)			0.32	0.13				
trimester 1			2000-2003	0.93 (0.78, 1.12)	IQR 5.2 ppb											
90 days before birth				1.03 (0.86, 1.23)	IQR 4.5 ppb											

TABLE AX6.6-2 (cont'd). PRETERM DELIVERY AND LONG-TERM NO_2 EXPOSURE STUDIES

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb) Low	Mid-range	High	Correlation with Other Pollutants $PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
Leem et al. (2006)	Inchon, Korea	Preterm	52,113 birth cert		Log binomial		15.78	22.93	29.9		0.37		0.54	0.63		Kriging
Trimester 1 Q2			2001-2002	1.13 (0.99, 1.27)	regression											
Trimester 1 Q3				1.07 (0.94, 1.21)												
Trimester 1 Q4				1.24 (1.09, 1.41)	Trend .02											
Trimester 3 Q2				1.06 (0.93, 1.20)												
Trimester 3 Q3				1.14 (1.01, 1.29)												
Trimester 3 Q4				1.21 (1.07, 1.37)	Trend <.001											

TABLE AX6.6-3. FETAL GROWTH AND LONG-TERM NO_2 EXPOSURE STUDIES

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
Salam et al. (2005)	Southern CA	Term SGA	3,901 birth cert		Linear mixed model	24 h		36.1 (15.4)			0.55	−0.1		0.69		5 km or 3 monitors within 50 km
Pregnancy	CHS	<15% of data	1975-1987	1.1 (0.9, 1.3)												
Trimester 1				1.2 (1.0, 1.4)	IQR = 25											
Trimester 2				1.0 (0.8, 1.2)												
Trimester 3				1.0 (0.8, 1.2)												
Mannes et al. (2005)	Sydney	SGA	51,460 birth cert		Logistic regression	1-h max	18	23	27.5	0.66	0.47	0.29		0.57		5 km
Trimester 1		>2sd below	1998-2000	1.06 (0.99, 1.14)				23.2 (7.4)								
Trimester2		national data		1.14 (1.07, 1.22)	1 ppb											
Trimester 3				1.13 (1.05, 1.21)												
1 mo before birth				1.07 (1.00, 1.14)												
Liu et al. (2003)	Vancouver	term SGA	229,085 birth cert		Logistic regression	24 h	15.1	18.1	22.3			−0.25	0.61	0.72		13 monitors Avg
Trimester 1		<10% national	1986-1998	1.03 (0.98, 1.10)												
Trimester 2				0.94 (0.88, 1.00)	10 ppb											
Trimester 3				0.98 (0.92, 1.06)												
First mo				1.05 (1.01, 1.10)												
Last mo				0.98 (0.92, 1.03)												

TABLE AX6.7-1. LUNG FUNCTION AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants					BS	Distance
							Low	Mid-Range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO		
Gauderman (2004)	southern CA	Lung function	1757 children		2-stage linear	24 h annual				0.79	0.67	−0.11				Study monitors in 12 towns
Difference in lung growth		Longitudinal	age 10-18		Regression	Avg										
FVC			CHS	−95 (−183.4, −0.6)												
FEV_1				−101.4 (−164.5, −38.4)	34.6 ppb											
MMEF				−211 (−377.6, −44.4)												
Moseler et al. (1994)	Frieberg	lung function	467 children		Linear regression	Median wkly		21.28								
with asthma symp	Germany		age 9-16					threshold								
FEV_1				0.437	Parameter estimates											
lnMEF75%				−0.011												
ln MEF50%				−0.022	µg											
lnMEF25%				−0.029												
no asthma symp																
FEV_1				−0.049												
lnMEF75%				0.003												
lnMEF50%				0.004												
lnMEF25%				0.003												
Ackermann-Liebrich et al. (1997)	Switzerland	Lung function	3,115 adults		2-stage linear	24-h annual		18.9 (8.5)			0.91	−0.78	0.86			Sonitors in 8 Study area
			3 yr residents		Regression	Avg										
FVC			nonsmokers	−0.0123												
			SAPALDIA	(−0.0152, −0.0094)	Parameter estimate											
FEV_1				−0.0070												
				(−0.0099, −0.0041)	10 µg											

TABLE AX6.7-1 (cont'd). LUNG FUNCTION AND LONG-TERM NO_2 EFFECTS

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants					BS	Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO		
Schindler et al. (1998)	Switzerland	Lung function	560 adults	%change	Linear regression	Wkly avg										Personal and Home monitors
FVC home			3 yr residents	−0.59 (−1)												
FVC personal			SAPALDIA													
FEV home																
FEV personal																
Peters et al. (1999a)	Southern CA	Lung function	3,293 children		Linear regression	24 h										Study monitors in 12 tons
			CHS													
FVC all 1986-1990				−42.6 (13.5)	Parameter estimates											
FVC girls 1986-1990				−58.5 (15.4)	IQR = 25 ppb											
FEV_1 all 1986-1990				−23.2 (12.5)												
FEV_1 girls 1986-1990				−39.9 (13.9)	IQR = 25 ppb											
FVC all 1994				−46.2 (16.0)												
FVC girls 1994				−56.7 (19.8)												
FEV_1 all.1994				−22.3 (14.8)												
FEV_1 girls 1994				−44.1 (16.1)												
Tager et al. (2005)	Southern & Northern CA	Lung function	255 students UC Berkeley		Linear regression		22	30	40	Men		0.57				Lifetime history
lnFEF75 men				−0.029 (0.003)	Parameter estimates		21	27	40	Women						
lnFEF75 women				−0.032 (0.002)	Results substantially											

TABLE AX6.7-2. ASTHMA AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% I)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants					BS	Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO		
Garrett et al. (1999)	Latrobe Valley	Asthma	148 children		Logistic regression			6								In home
Bedroom NO_2	Australia	Monash Q	Age 7-14	1.01 (0.75, 1.37)												
Indoor mean			1994-1995	1.00 (.075, 1.31)	10 µg											
winter				0.99 (0.84, 1.16)												
summer				2.52 (0.99, 6.42)												
																4 monitors
Hirsch et al. (1999)	Dresden	Asthma	5,421 children		Logistic regression		29.3	33.8	37.8							Within 1 km
Home address	Germany	ISAAC	Age 5-7, 9-11	1.16 (0.94, 1.42)												
Home & school			1995-1996	1.14 (0.86, 1.51)	10 µg											
			12 mo residence													
Peters et al. (1999b)	Southern CA	Asthma	3,676 children		Logistic regression	24 h		21.5 mean								Study monitors
all children	CHS	Questionnaire	Age 9-16	1.21 (0.850, 1.71)												In 12 towns
boys			1994	1.25 (0.90, 1.75)	IQR = 25 ppb											
girls				1.07 (0.57, 2.02)												

TABLE AX6.7-2 (cont'd). ASTHMA AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% I)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
Millstein et al. (2004)	Southern CA	Asthma	2,034 children		Mixed effects model	Moly				0.28	0.39					Study monitors in 12 towns
annual	CHS	Medication use	Age 9-11	0.94 (0.71, 1.22)												
March-August			1995	0.96 (0.68, 1.37)	IQR = 5.74 ppb											
Sept-Feb				0.90 (0.66, 1.24)												
Penard-Morand et al. (2005)	France 6 towns	Asthma ISAAC	4,901 children Age 9-11		Logistic regression	3 yrs										29 monitoring Sites, school
lifetime asthma			1999-2000	0.94 (0.83, 1.07)	10 µg		8.7, 16.0		16.1, 25.7		0.46	0.76	0.35			Address
current asthma			3 yr residence	0.92 (0.77, 1.10)												
Studnicka et al. (1997)	8 communities	Asthma	843 children		Logistic regression	3 yrs	8.0, 8.7	11.7, 13.3	14.7, 17.0							Study monitor in each community
Ever asthma low	Lower Austria	ISAAC		1.28												
Ever asthma medium				2.14												
Ever asthma high				5.81	<.05											
Current asthma low				1.7												
Currrent asthma medium				1.47												
Current asthma high				8.78	<.05											

TABLE AX6.7-2 (cont'd). ASTHMA AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% I)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	
Wang et al. (1999)	Taiwan	Asthma	117,080 students		Logistic regression			28 median								24 district monitors
Current asthma			age 11-16	1.08 (1.04, 1.13)	Above/below median											
Ramadour M et al. (2000)	7 communities	Asthma	2,445 children		Logisitic regression			11-27 mean								Monitors in each community
	France	ISAAC	age 13-14	Nonsignificant												
			3 yr residence	Results												
Shima and Adachi et al. (2000)	7 communities	Asthma	905 children		Logistic regression		20-29	30-39	≥40							In home measurements
Outdoor 4th grade girls	Japan	Prevalence	age 9-10	1.14 (0.65, 2.09)												
Outdoor 5th grade girls				1.14 (0.63, 2.13)	10 ppb increase			7-25 mean								Monitors near schools
Outdoor 6th grade girls				0.95 (0.45, 2.05)				Outdoors								
Indoor 4th grade girls				1.63 (1.06, 2.54)												
Indoor 5th grade girls				1.67 (1.06, 2.66)												
Indoor 6th grade girls				1.18 (0.62, 2.18)												
Outdoor		Asthma		2.10 (1.10, 4.75)	10 ppb increase											
Indoor		Incidence		0.87 (0.51, 1.43)												

TABLE AX6.7-2 (cont'd). ASTHMA AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% I)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants					BS	Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO		
Kim J et al. (2004a)	San Francisco	Asthma	1,109 children		2-stage			24 mean		"low"		"low"				10 school sites
All children	Bay area		Age 9-11	1.02 (0.97, 1.07)	Hierarchical model											
All 1 yr residents				1.04 (0.98, 1.10)												
1 yr resident girls				1.09 (1.03, 1.15)	IQR = 3.6 NO_2											
1 yr resident boys				1.00 (0.94, 1.07)												
All children				1.04 (0.97, 1.11)												
All 1 yr residents				1.07 (1.00, 1.14)	IQR = 14.9 NO_x											
1 yr resident girls				1.17 (1.06, 1.29)												
1 yr resident boys				1.02 (0.93, 1.11)												
Gauderman W et al. (2005)	Southern CA	Asthma	208 children		Logistic regression	4 wk avg		13-51								Outside home
Lifetime asthma	CHS			1.83 (1.04, 3.21)	IQR = 5.7											
Asthma med use				2.19 (1.20, 4.01)												
Hwang et al. (2005)	Taiwan	Asthma	32,672 children		2-stage		21.5	29.6	33.1		0.34	−0.39	0.5			Schools within
Parental atopy	National study	ISAAC		0.99 (0.92, 1.07)	Hierarchical model											1 km of monitors
No parental atopy				1.02 (0.95, 1.10)	10 ppb NO_x											

TABLE AX6.7-3. RESPIRATORY SYMPTOMS AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
Garrett et al. (1999)	Latrobe Valley	Symptoms	148 children		Logistic regression			6								In home
wheeze	Australia	Monash Q	Age 7-14	1.15 (0.85, 1.54)												
cough			1994-1995	1.47 (0.99, 2.18)	10 µg											
short of breath				1.23 (0.92, 1.64)												
chest tightness				1.12 (0.81, 1.56)												
any symptoms				1.24 (0.91, 1.68)	10 µg mean											4 monitors
any symptoms				1.12 (0.93, 1.35)	10 µg winter											
any symptoms				2.71 (1.11, 6.59)	10 µg summer											
Hirsch et al. (1999)	Dresden	Symptoms	5,421 children		Logistic regression		29.3	33.8	37.8							Within 1 km
wheeze home	Germany	ISAAC	Age 5-7, 9-11	1.13 (0.93, 1.37)												
wheeze school			1995-1996	0.95 (0.72, 1.26)	10 µg											
cough home			12 mo residence	1.22 (1.94, 1.44)												
cough school				1.21 (0.96, 1.52)												
cough non-atopic child				1.42 (1.10, 1.84)												
Peters et al. (1999b)	Southern CA	Symptoms	3,676 children		Logistic regression	24 h		21.5 mean								Study monitors
wheeze	CHS	Questionnaire	Age 9-16	1.12 (0.86, 1.45)												In 12 towns
cough			1994	1.14 (0.94, 1.39)	IQR = 25 ppb											
wheeze boys				1.54 (1.04, −2.29)												
wheeze girls				0.86 (0.57, 1.29)												

TABLE AX6.7-3 (cont'd). RESPIRATORY SYMPTOMS AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						Distance
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	
Millstein et al. (2004)	Southern CA	Symptoms	2,034 children		Mixed effects model	Moly				0.28	0.39					Study monitors
wheeze	CHS		Age 9-11	0.93 (0.77, 1.12)												In 12 towns
wheeze Mar-Aug			1995	0.79 (0.40, 1.53)	IQR = 5.74 ppb											
wheeze Sept-Feb				0.85 (0.64, 1.14)												
Penard-Morand et al. (2005)	France 6 towns	Symptoms	4,901 children		Logistic regression	3 yrs										29 monitoring sites, school address
		ISSAC	Age 9-11													
wheeze past 12 mos.			1999-2000	0.87 (0.75, 1.01)	10 μg		8.7, 16.0		16.1, 25.7		0.46	0.76	0.35			
			3 yr residence													
Mukala et al. (1999)	Helsinki	Symptoms	163 children		GEE	Wkly	<8.6	8.6, 14.5	>14.5							Palms tubes
cough	Finland		Age 3-6	1.23 (0.89, 1.70)	2nd tertile	Avg										On outer garment
cough			1991	1.52 (1.00, 2.31)	3rd tertile											
nasal symp winter				0.99 (0.58, 1.68)	2nd tertile											
nasal symp winter				0.89 (0.44. 1.82)	3rd tertile											
nasal symp spring				0.76 (0.56, 1.02)	2nd tertile											
nasal symp spring				0.68 (0.46, 1.01)	3rd tertile											
Pikhart et al. (2000)	Prague	Symptoms	3,045 children		Multi-level model		14.8	19	24.1							
wheeze	Czech	SAVIAH	Age 7-10	1.16 (0.95, 1.42)	Individual covariates											
wheeze			1993-1994	1.07 (0.86, 1.33)	Ecological covariates											
wheeze				1.08 (0.86, 1.36)	Both covariates											

TABLE AX6.7-3 (cont'd). RESPIRATORY SYMPTOMS AND LONG-TERM NO_2 EXPOSURE

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	Distance
van Strien, 2004	CT and MA	Symptoms	849 children		Poisson regression	10-14 day	5.1	9.9	17.4							In home
wheeze			Age 12 mos	1.15 (0.79, 1.67)	Q2	Avg										
wheeze				1.03 (0.69, 1.53)	Q3											
wheeze				1.45 (0.92, 2.27)	Q4											
cough				0.96 (0.69, 1.36)	Q2											
cough				1.33 (0.94, 1.88)	Q3											
cough				1.52 (1.00, 2.31)	Q4											
short of breath				1.59 (0.96, 2.62)	Q2											
short of breath				1.95 (1.17. 3.27)	Q3											
short of breath				2.38 (1.31, 4.34)	Q4											
Nitschke et al. (2006)	Adelaide	Symptoms	174 asthmatic		Zero-inflated negative		School	34 (28)	117 max							9 days in class
Wheeze school	Australia		Children, age 5-13	0.99 (0.93, 1.06)	binomial regression		Home	20 (22)	147 max							3 days at home
Wheeze home			2000	1.00 (0.90, 1.11)												
Cough school				1.01 (0.98, 1.04)	10ppb increase											
Cough home				0.99 (0.96, 1.02)												
Difficult breath school				1.11 (1.05, 1.18)												
Difficult breath home				1.03 (1.01, 1.05)												
Chest tight school				1.12 (1.07, 1.17)												
Chest tight home				1.02 (0.95, 1.09)												

TABLE AX6.8. LUNG CANCER

Author, Year	Study Location	Study Group	Study Subjects	Odds Ratio (95% CI)	Analysis Method	Unit of Averaging Time	Conc Range (ppb)			Correlation with Other Pollutants						Exposure
							Low	Mid-range	High	$PM_{2.5}$	PM_{10}	O_3	SO_2	CO	BS	
Nyberg et al. (2000)	Stockholm	lung cancer	1,042 cases		logistic regression		8.1	10.6	13.3							From addresses
30-yr estimated exposure	Sweden		2,364 controls	1.05 (0.93, 1.18)	10 µg											and traffic
			men age 40-75	1.18 (0.93, 1.49)	Q2											
				0.90 (0.71, 1.14)	Q3											
				1.05 (0.79, 1.40)	Q4											
10-yr estimated exposure				1.10 (0.97, 1.23)	10 µg											
				1.15 (0.91, 1.46)	Q2											
				1.01 (0.79, 1.29)	Q3											
				1.07 (0.81, 1.42)	Q4											
				1.44 (1.05, 1.99)	90th percentile											
Nafstad (2004)	Norway	lung caner	16,209 men		Cox proportional		5.32	10.6	16							Home address
lung cancer incidence			age 40-49 at entry	1.08 (1.02, 1.15)	10 µg											1972-1974
			followed 1972-1998	0.90 (0.70, 1.15)	Q2											
				1.06 (0.81, 1.38)	Q3											
				1.36 (1.01, 1.83)	Q4											
non-lung cancer				1.02 (0.99, 1.06	10 µg											
				0.98 (0.88, 1.08)	Q2											
				1.05 (0.94, 1.18)	Q3											
				1.04 (0.91, 1.18)	Q4											

TABLE AX6.9. EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
META ANALYSIS						
Stieb et al. (2002), re-analysis (2003) meta-analysis of estimates from multiple countries.	All cause	24-h avg ranged from 13 ppb (Brisbane, Australia) to 38 ppb (Santiago, Chile). "Representative" concentration: 24 ppb	PM_{10}, O_3, SO_2, CO	The lags and multiday averaging used in these estimates varied	Meta-analysis of time-series study results	Single-pollutant model (11 estimates): 0.8% (95% CI: 0.2, 1.5); Multipollutant model estimates (3 estimates): 0.4% (95% CI: −0.2, 1.1)
UNITED STATES						
Samet et al. (2000a,b reanalysis Dominici et al., 2003) 90 U.S. cities (58 U.S. cities with NO_2 data) 1987-1994	All cause; cardiopulmonary	Ranged from 9 ppb (Kansas City) to 39 ppb (Los Angeles), 24-h avg	PM_{10}, O_3, SO_2, CO; two-pollutant models	0, 1, 2	Poisson GAM, reanalyzed with stringent convergence criteria; Poisson GLM. Time-series study.	24-h avg NO_2 (per 20 ppb): Posterior means: All cause: Lag 1: 0.50% (0.09, 0.90) Lag 1 with PM_{10} and SO_2: 0.48% (−0.54, 1.51)
Kinney and Özkaynak (1991) Los Angeles County, CA 1970-1979	All cause; respiratory; circulatory	69 ppb, 24-h avg	KM (particle optical reflectance), NO_2, SO_2, CO; multipollutant models	1	OLS (ordinary least squares) on high-pass filtered variables. Time-series study.	All cause: Exhaustive multipollutant model: 0.5% (−0.1, 1.2); Two-pollutant with O_x: 0.7% (0.5, 1.0)
Kelsall et al. (1997) Philadelphia, PA, 1974-1988	All cause; respiratory; cardiovascular,	39.6 ppb, 24-h avg	TSP, CO, SO_2, O_3	0 (AIC presented for 0 through 5)	Poisson GAM	All cause: Single pollutant: 0.3% (−0.6, 1.1); With TSP: −1.2% (−2.2, −0.2)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
UNITED STATES (cont'd)						
Ostro et al. (2000) Coachella Valley, CA 1989-1998	All cause; respiratory; cardiovascular; cancer; other	20 ppb, 24-h avg	PM_{10}, $PM_{2.5}$, $PM_{10-2.5}$, O_3, CO	0-4	Poisson GAM with default convergence criteria. Time-series study.	Lag 0 day: All cause: 5.5% (1.0, 10.3) Respiratory: 1.8% (−10.3, 15.5) Cardiovascular: 3.7% (−1.7, 9.3)
Fairley (1999; reanalysis Fairley, 2003) Santa Clara County, CA 1989-1996	All cause; respiratory; circulatory	28 ppb, 24-h avg	PM_{10}, $PM_{2.5}$, $PM_{10-2.5}$, SO_4^{2-}, coefficient of haze, NO_3^-, O_3, SO_2;	0, 1	Poisson GAM, reanalyzed with stringent convergence criteria; Poisson GLM. Time-series study.	Lag 1: All cause: 1.9% (0.2, 3.7); Cardiovascular: 1.4% (−1.7, 4.5); Respiratory: 4.8% (−0.3, 10.2)
Gamble (1998) Dallas, TX 1990-1994	All cause; cardiopulmonary	15 ppb, 24-h avg	PM_{10}, O_3, SO_2, CO; two-pollutant models	Avg 4-5	Poisson GLM. Time-series study.	All cause: 4.4% (0.0, 9.0) Cardiovascular: 1.9% (−4.6, 9.0) Respiratory: 13.7% (−2.0, 32.0)
Dockery et al. (1992) St. Louis, MO and Eastern Tennessee 1985-1986	All cause	St. Louis: 20 ppb; Eastern Tennessee: 12.6 ppb, 24-h avg	PM_{10}, $PM_{2.5}$, SO_4, H^+, O_3, SO_2	Lag 1	Poisson with GEE. Time-series study.	All cause: St. Louis, MO: 0.7% (−3.5, 5.1) Eastern Tennessee: 3.9% (−8.7, 18.2)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
UNITED STATES (cont'd)						
Moolgavkar (2003) Cook County, IL and Los Angeles County, CA, 1987-1995	All cause; cardiovascular	Cook County: 25 ppb; Los Angeles: 38 ppb, 24-h avg	$PM_{2.5}$, PM_{10}, O_3, SO_2, CO; two-pollutant models	0, 1, 2, 3, 4, 5	Poisson GAM with default convergence criteria. Time-series study.	All cause: Lag 1: Cook County: Single pollutant: 2.2% (1.3, 3.1); with PM_{10}: 1.8% (0.7, 3.0); Los Angeles: Single pollutant: 2.0% (1.6, 2.5); with $PM_{2.5}$: 1.8% (0.1, 3.6).
Moolgavkar (2000a,b,c); Re-analysis (2003). Cook County, IL; Los Angeles County, CA; and Maricopa County, AZ, 1987-1995	Cardiovascular; cerebrovascular; COPD	Cook County: 25 ppb; Los Angeles: 38 ppb; Maricopa County: 19 ppb, 24-h avg	$PM_{2.5}$, PM_{10}, O_3, SO_2, CO; two- and three-pollutant models	0, 1, 2, 3, 4, 5	Poisson GAM with default convergence criteria in the original Moolgavkar (2000); GAM with stringent convergence criteria and GLM with natural splines in the 2003 re-analysis. The 2000 analysis presented total death risk estimates only in figures.	GAM, Lag 1: Cardiovascular: Cook County: 1.1% (−0.5, 2.8); Los Angeles: 2.8% (2.0, 3.6); Maricopa Co.: 4.6% (0.5, 9.0); Re-analysis, GLM: Total deaths: 2.5% (1.5, 3.6)
Lippmann et al. (2000; reanalysis Ito, 2003, 2004) Detroit, MI 1985-1990 1992-1994	All cause; respiratory; circulatory; cause-specific	1985-1990: 23.3 ppb, 24-h avg 1992-1994: 21.3 ppb, 24-h avg	PM_{10}, $PM_{2.5}$, $PM_{10-2.5}$, SO_4^{2-}, H^+, O_3, SO_2, CO; two-pollutant models	0, 1, 2, 3, 0-1, 0-2, 0-3	Poisson GAM, reanalyzed with stringent convergence criteria; Poisson GLM. Numerical NO_2 risk estimates were not presented in the re-analysis. Time-series study.	Poisson GAM: All cause: Lag 1: 1985-1990: 0.9% (−1.2, 3.0) 1992-1994: 1.3% (−1.5, 4.2)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
UNITED STATES (cont'd)						
Lipfert et al. (2000a) Seven counties in Philadelphia, PA area 1991-1995	All cause; respiratory; cardiovascular; all ages; age 65+ yrs; age <65 yrs; various subregional boundaries	20.4 ppb, 24-h avg	PM_{10}, $PM_{2.5}$, $PM_{10-2.5}$, SO_4 O_3, other PM indices, NO_2, SO_2, CO; two-pollutant models	0-1	Linear with 19-day weighted avg Shumway filters. Time-series study. Numerous results.	All-cause, avg of 0- and 1-day lags, Philadelphia: 2.2% ($p > 0.05$)
Chock et al. (2000) Pittsburgh, PA 1989-1991	All cause; age <74 yrs; age 75+ yrs	Not reported.	PM_{10}, NO_2, SO_2, CO; two-, five-, and six-pollutant models	0, plus minus 3 days.	Poisson GLM. Time-series study. Numerous results	All cause, lag 0, age 0-74: 0.5% (−2.4, 3.5); age 75+: 1.0% (−1.9, 4.0).

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
De Leon et al. (2003) New York City 1985-1994	Circulatory and cancer with and without contributing respiratory causes	40.6 ppb, 24-h avg	PM_{10}, O_3, SO_2, CO; two-pollutant models	0 or 1	Poisson GAM with stringent convergence criteria; Poisson GLM. Time-series study.	Gaseous pollutants results were given only in figures. Circulatory: Age < 75: ~1% Age 75+ : ~ 2%
Klemm and Mason (2000); Klemm et al. (2004) Atlanta, GA Aug 1998-July 2000	All cause; respiratory; cardiovascular; cancer; other; age <65 yrs; age 65+ yrs	51.3 ppb, max 1-h.	$PM_{2.5}$, $PM_{10-2.5}$, EC, OC, O_3, SO_4^{2-}, NO_3^-, SO_2, CO	0-1	Poisson GLM using quarterly, moly, or biweekly knots for temporal smoothing. Time-series study.	All cause, age 65+ yrs: avg 0-1 days Quarterly knots: 1.0% (−4.2,6.6); Moly knots: 3.1% (−3.0, 9.7); Bi-wkly knots: 0.9% (−5.9, 8.2).
Gwynn et al. (2000) Buffalo, NY	All cause; respiratory; circulatory	24-h avg 21 ppb	PM_{10}, CoH, O_3, SO_2, CO, H^+, SO_4^{2-}		Poisson GAM with Default convergence criteria. Time-series study.	All cause (lag 3): 2.1% (−0.3, 4.6); Circulatory (lag 2): 1.3% (−2.9, 5.6); Respiratory (lag 1): 6.4% (−2.5, 16.2)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
CANADA						
Burnett et al. (2004) 12 Canadian cities 1981-1999	All cause	24-h avg ranged from 10 (Saint John) to 26 (Calgary) ppb.	$PM_{2.5}$, $PM_{10-2.5}$, O_3, SO_2, CO	1, 0-2	Poisson GLM. Time-series study.	Lag 0-2, single pollutant: 2.0% (1.1, 2.9); with O_3: 1.8% (0.9, 2.7). Days when PM indices available, lag 1, single pollutant: 2.4% (0.7, 4.1); with $PM_{2.5}$: 3.1% (1.2, 5.1).
Burnett et al. (2000), re-analysis (2003) 8 Canadian cities 1986-1996	All cause	24-h avg ranged from 15 (Winnipeg) to 26 (Calgary) ppb.	$PM_{2.5}$, PM_{10}, $PM_{2.5-10}$, SO_2, O_3, CO	0, 1, 0-2	Poisson GAM with default convergence criteria. Time-series study. The 2003 re-analysis did not consider gaseous pollutants.	Days when PM indices available, lag 1, single pollutant: 3.6% (1.6, 5.7); with $PM_{2.5}$: 2.8% (0.5, 5.2).
Burnett et al. (1998a), 11 Canadian cities 1980-1991	All cause	24-h avg ranged from 14 (Winnipeg) to 28 (Calgary) ppb.	SO_2, O_3, CO	0, 1, 2, 0-1, 0-2 examined but the best lag/averaging for each city chosen	Poisson GAM with default convergence criteria. Time-series study.	Single pollutant: 4.5% (3.0, 6.0); with all gaseous pollutants: 3.5% (1.7, 5.3).
Burnett et al. (1998b), Toronto, 1980-1994	All cause	24-h avg 25 ppb.	SO_2, O_3, CO, TSP, COH, estimated PM_{10}, estimated $PM_{2.5}$	0, 1, 0-1	Poisson GAM with default convergence criteria. Time-series study.	Single pollutant (lag 0): 1.7% (0.7, 2.7); with CO: 0.4% (−0.6, 1.5).

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
CANADA (cont'd)						
Vedal et al. (2003) Vancouver, British Columbia, Canada 1994-1996	All cause; respiratory; cardiovascular	17 ppb, 24-h avg	PM_{10}, O_3, SO_2, CO	0, 1, 2	Poisson GAM with stringent convergence criteria. Time-series study. By season.	Results presented in figures only. NO_2 showed associations in winter but not in summer.
Villeneuve et al. (2003) Vancouver, British Columbia, Canada 1986-1999	All cause; respiratory; cardiovascular; cancer; socioeconomic status	19 ppb, 24-h avg	$PM_{2.5}$, PM_{10}, $PM_{2.5-10}$, TSP, coefficient of haze, SO_4^{2-}, SO_2, O_3, CO	0, 1, 0-2	Poisson GLM with natural splines. Time-series study.	All yr: All cause Lag 1: 4.0% (0.9, 7.2) Respiratory: Lag 0: 2.1% (−3.0, 7.4) Cardiovascular: Lag 0: 4.3% (−4.2, 13.4)
Goldberg et al. (2003) Montreal, Quebec, Canada 1984-1993	Congestive heart Failure (CHF) as underlying cause of death vs. those classified as having congestive heart failure 1 yr prior to death	22 ppb, 24-h avg	$PM_{2.5}$, coefficient of haze, SO_4^{2-}, SO_2, O_3, CO	0, 1, 0-2	Poisson GLM with natural splines. Time-series study.	CHF as underlying cause of death: Lag 1: 1.0% (−5.1, 7.5) Having CHF 1 yr prior to death: Lag 1: 3.4% (0.9, 6.0)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE						
Samoli et al. (2006) 30 APHEA2 cities. Study periods vary by city, ranging from 1990 to 1997	All cause, respiratory; cardiovascular	1-h max ranged from 24 (Wroclaw) to 81 (Milan) ppb	BS, PM_{10}, SO_2, O_3	01	Poisson model with penalized splines.	All-cause: single: 1.8% (1.3, 2.2); with SO_2: 1.5% (1.0, 2.0) Cardiovascular: single: 2.3% (1.7, 3.0); with SO_2: 1.9% (1.1, 2.7) Respiratory: single: 2.2% (1.0, 3.4); with SO_2: 1.1% (−0.4, 2.6)
Samoli et al. (2005) 9 APHEA2 cities. Period not reported.	All-cause	The selected cities had 1-h max medians above 58 ppb and the third quartiles above 68.	None	01	Poisson model with either non-parametric or cubic spline smooth function in each city, and combined across cities.	No numeric estimate presented. The concentration-response was approximately linear.
Touloumi et al. (1997) Six European cities: London, Paris, Lyon, Barcelona, Athens, Koln. Study periods vary by city, ranging from 1977 to 1992	All cause	Ranged from 37 (Paris) to 70 (Athens) ppb, 1-h max	BS, O_3; two-pollutant models	0, 1, 2, 3, 0-1, 0-2, 0-3 (best lag selected for each city)	Poisson autoregressive. Time-series study.	All-cause: Single-pollutant model: 1.0% (0.6, 1.3); With BS: 0.5% (0.0, 0.9).
Zmirou et al. (1998) Four European cities: London, Paris, Lyon, Barcelona Study periods vary by city, ranging from 1985-1992	Respiratory; cardiovascular	Ranged from 24 (Paris) to 37 (Athens) ppb in cold season and 23 (Paris) to 37 (Athens) ppb in warm season, 24-h avg	BS, TSP, SO_2, O_3	0, 1, 2, 3, 0-1, 0-2, 0-3 (best lag selected for each city)	Poisson GLM. Time-series study.	Western Europe: Respiratory: 0.0% (−1.1, 1.1) Cardiovascular: 0.8% (0.0, 1.5)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Biggeri et al. (2005) 8 Italian cities, Period variable between 1990-1999	All cause; respiratory; cardiovascular	24-h avg ranged from 30 (Verona) to 51 (Rome) ppb	Only single-pollutant models; O_3, SO_2, CO, PM_{10}	0-1	Poisson GLM. Time-series study.	All cause: 3.6% (2.3, 5.0); Respiratory: 5.6% (0.2, 11.2) Cardiovascular: 5.1% (3.0, 7.3)
Anderson et al. (1996) London, England 1987-1992	All cause; respiratory; cardiovascular	37 ppb, 24-h avg	BS, O_3, SO_2; two-pollutant models	0, 1	Poisson GLM. Time-series study.	All cause (Lag 1): 0.6% (−0.1, 1.2); Respiratory (lag 1): −0.7% (−2.3, 1.0) Cardiovascular: 0.5% (−0.4, 1.4)
Bremner et al. (1999) London, England 1992-1994	All cause; respiratory; cardiovascular; all cancer; all others; all ages; age specific (0-64, 65+, 65-74, 75+ yrs)	34 ppb, 24-h avg	BS, PM_{10}, O_3, SO_2, CO; two-pollutant models	Selected best from 0, 1, 2, 3, (all cause); 0, 1, 2, 3, 0-1, 0-2, 0-3 (respiratory, cardiovascular)	Poisson GLM. Time-series study.	All cause (lag 1): 0.9% (0.0, 1.9) Respiratory (lag 3): 1.9% (−0.3, 4.2) Cardiovascular (lag 1): 1.9% (0.6, 3.2)
Anderson et al. (2001) West Midlands region, England 1994-1996	All cause; respiratory; cardiovascular.	37 ppb, 1-h max	PM_{10}, $PM_{2.5}$, $PM_{2.5-10}$, BS, SO_4^{2-}, O_3, SO_2, CO	0-1	Poisson GAM with default convergence criteria. Time-series study.	All cause: 1.7% (−0.5, 3.8) Respiratory: 3.3% (−1.9, 8.8) Cardiovascular: 3.1% (−0.2, 6.4)
Prescott et al. (1998) Edinburgh, Scotland 1992-1995	All cause; respiratory; cardiovascular; all ages; age <65 yrs; age ≥65 yrs	26 ppb, 24-h avg	BS, PM_{10}, O_3, SO_2, CO; two-pollutant models	0	Poisson GLM. Time-series study.	Results presented as figures only. Essentially no associations in all categories. Very wide confidence intervals.

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Le Tertre et al. (2002a) Le Havre, Lyon, Paris, Rouen, Strasbourg, and Toulouse, France Study periods vary by city, ranging from 1990-1995	All cause; respiratory; cardiovascular	Ranged from 15 (Toulouse) to 28 (Paris) ppb, 24-h avg	BS, O_3, SO_2	0-1	Poisson GAM with default convergence criteria. Time-series study.	Six-city pooled estimates: All cause: 2.9% (1.6, 4.2) Respiratory: 3.1% (−1.7, 8.0) Cardiovascular: 3.5% (1.1, 5.9)
Zeghnoun et al. (2001) Rouen and Le Havre, France 1990-1995	All cause; respiratory; cardiovascular	24-h avg 18 ppb in Rouen; 20 ppb in Le Havre	SO_2, BS, PM_{13}, O_3	0, 1, 2, 3, 0-3,	Poisson GAM with default convergence criteria. Time-series study.	All cause in Rouen (lag 1): 5.5% (0.2, 11.1) ; in Le Havre (lag 1): 2.4% (−3.4, 8.5)
Dab et al. (1996) Paris, France 1987-1992	Respiratory	24 ppb, 24-h avg	BS, PM_{13}, O_3, SO_2, CO	0	Poisson autoregressive. Time-series study.	Lag1: 2.1% (3.1, 7.7)
Zmirou et al. (1996) Lyon, France 1985-1990	All cause; respiratory; cardiovascular; digestive	37 ppb, 24-h avg	PM_{13}, SO_2, O_3	Selected best from 0, 1, 2, 3	Poisson GLM. Time-series study.	All cause (lag 1): 1.5% (−1.5, 4.6) Respiratory (lag 2): −2.3% (−15.6, 13.0) Cardiovascular (lag 1): 0.8% (−2.7, 4.3)
Sartor et al. (1995) Belgium Summer 1994	All cause; age <65 yrs; age 65+ yrs	24-h avg NO_2: Geometric mean: During heat wave (42-day period): 17 ppb Before heat wave (43-day period): 15 ppb After heat wave (39-day period): 13 ppb	TSP, NO, O_3, SO_2	0, 1, 2	Log-linear regression for O_3 and temperature. Time-series study.	Only correlation coefficients presented for NO_2. Unlike O_3, NO_2 was not particularly elevated during the heat wave.

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Hoek et al. (2000; reanalysis Hoek, 2003) The Netherlands: entire country, four urban areas 1986-1994	All cause; COPD; pneumonia; cardiovascular	24-h avg median: 17 ppb in the Netherlands; 24 ppb in the four major cities	PM_{10}, BS, SO_4^{2-}, NO_3^-, O_3, SO_2, CO; two-pollutant models	1, 0-6	Poisson GAM, reanalyzed with stringent convergence criteria; Poisson GLM. Time-series study.	Poisson GLM: All cause: Lag 1: 1.9% (1.2, 2.7) Lag 0-6: 2.6% (1.2, 4.0); with BS: 1.3% (−0.9, 3.5); Cardiovascular (lag 0-6): 2.7% (0.7, 4.7). COPD (lag 0-6): 10.4% (4.5, 16.7). Pneumonia (lag 0-6): 19.9% (11.5, 29.0).
Hoek et al. (2001; reanalysis Hoek, 2003) The Netherlands 1986-1994	Total cardiovascular; myocardial infarction; arrhythmia; heart failure; cerebrovascular; thrombosis-related	24-h avg median: 17 ppb in the Netherlands; 24 ppb in the four major cities	PM_{10}, O_3, SO_2, CO	1	Poisson GAM, reanalyzed with stringent convergence criteria; Poisson GLM. Time-series study.	Poisson GLM: Total cardiovascular: 2.7% (0.7, 4.7) Myocardial infarction: 0.3% (−2.6, 3.2) Arrhythmia: 1.7% (−6.6, 10.6) Heart failure: 7.6% (1.4, 14.2) Cerebrovascular: 5.1% (0.9, 9.6) Thrombosis-related: −1.2% (−9.6, 8.1)
Roemer and van Wijnen (2001) Amsterdam, the Netherlands 1987-1998	All cause	24-h avg: Background sites: 24 ppb Traffic sites: 34 ppb	BS, PM_{10}, O_3, SO_2, CO	1, 2, 0-6	Poisson GAM with default convergence criteria (only one smoother). Time-series study.	Total population using background sites: Lag 1: 3.8% (1.7, 5.9); Traffic pop. using background sites: lag 1: 5.7% (0.6, 11.0); Total pop. using traffic sites: Lag 1: 1.7% (0.4, 3.0)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Verhoeff et al. (1996) Amsterdam, the Netherlands 1986-1992	All cause; all ages; age 65+ yrs	1-h max O_3: 43 µg/m^3 Maximum 301	PM_{10}, O_3, CO; multipollutant models NO NO_2!!!	0, 1, 2	Poisson. Time-series study.	1-h max O_3 (per 100 µg/m^3) All ages: Lag 0: 1.8% (−3.8, 7.8) Lag 1: 0.1% (−4.7, 5.1) Lag 2: 4.9% (0.1, 10.0)
Fischer et al. (2003) The Netherlands, 1986-1994	All-cause, cardiovascular, COPD, and pneumonia in age groups <45, 45-64, 65-74, 75+	24-h avg median 17 ppb	PM_{10}, BS, O_3, SO_2, CO	0-6	Poisson GAM with default convergence criteria. Time-series study.	Cardiovascular: Age <45: −1.3% (−13.0, 12.1): age 45-64: −0.4% (−4.8, 4.3); age 65-74: 4.4% (0.8, 8.0); age 75 and up: 3.5% (1.4, 5.6)
Spix and Wichman (1996) Koln, Germany 1977-1985	All-cause	24-h avg 24 ppb; 1-h max 38 ppb	TSP, PM_7, SO_2	0, 1, 0-1	Poisson GLM. Time-series study.	Lag 1: 0.4% (−0.4, 1.2)
Peters et al. (2000b) NE Bavaria, Germany 1982-1994 Coal basin in Czech Republic 1993-1994	All cause; respiratory; cardiovascular; cancer	24-h avg: Czech Republic: 17.6 ppb Bavaria, Germany: 13.2 ppb	TSP, PM_{10}, O_3, SO_2, CO	0, 1, 2, 3	Poisson GLM. Time-series study.	Czech Republic: All cause: Lag 1: 2.1% (−1.7, 6.1) Bavaria, Germany: All cause: Lag 1: −0.1% (−3.6, 3.6)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Michelozzi et al. (1998) Rome, Italy 1992-1995	All-cause	24-h avg 52 ppb	PM_{13}, SO_2, O_3, CO	0, 1, 2, 3, 4	Poisson GAM with default convergence criteria. Time-series study.	Lag 2: all-yr: 1.6% (0.4, 2.9); cold season 0.3% (−1.2, 1.8); warm season: 4.2% (1.8,−6.6)
Pönkä et al. (1998) Helsinki, Finland 1987-1993	All cause; cardiovascular; age <65 yrs, age 65+ yrs	24-h avg: Median 20 ppb	TSP, PM_{10}, O_3, SO_2	0, 1, 2, 3, 4, 5, 6, 7	Poisson GLM. Time-series study.	No risk estimate presented for NO_2. PM_{10} and O_3 were reported to have stronger associations.
Saez et al. (2002) Seven Spanish cities, variable study periods between 1991 and 1996.	All cause; respiratory; cardiovascular	24-h avg mean ranged from 17 ppb in Huelva to 35 ppb in Valencia.	O_3, PM, SO_2, CO	0-3	Poisson GAM with default convergence criteria. Time-series study.	All cause: 2.6% (1.6, 3.6); with all other poll.: 1.7% (0.0, 3.3); Respiratory: 7.1% (−14.0, 33.5) Cardiovascular: 4.4% (−0.2, 9.2)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Garcia-Aymerich et al. (2000) Barcelona, Spain 1985-1989	All cause; respiratory; cardiovascular; general population; patients with COPD	Levels not reported.	BS, O_3, SO_2	Selected best avg lag	Poisson GLM. Time-series study.	All cause: General population: Lag 0-3: 3.3% (0.8, 5.8) COPD patients: Lag 0-2: 10.9% (0.4, 22.6) Respiratory: General population: Lag 0-1: 3.3% (−2.3, 9.2) COPD patients: Lag 0-2: 12.1% (−4.3, 31.4) Cardiovascular: General population: Lag 0-3: 2.4% (−0.9, 5.8) COPD patients: Lag 0-2: 4.3% (−13.6, 25.8)
Saez et al. (1999) Barcelona, Spain 1986-1989	Asthma mortality; age 2-45 yrs	Levels not reported.	BS, O_3, SO_2	0-2	Poisson with GEE. Time-series study.	RR = 4.1 (0.5, 35.0)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Sunyer et al. (1996) Barcelona, Spain 1985-1991	All cause; respiratory; cardiovascular; all ages; age 70+ yrs	1-h max: Median: Summer: 51 ppb Winter: 46 ppb	BS, SO_2, O_3	Selected best single-day lag	Autoregressive Poisson. Time-series study.	All yr, all ages: All cause: Lag 1: 1.9% (0.8, 3.1) Respiratory: Lag 0: 1.5% (−1.9, 5.0) Cardiovascular: Lag 1: 2.2% (0.5, 3.9) Summer risk estimates larger than winter risk estimates.
Sunyer and Basagãna (2001) Barcelona, Spain 1990-1995	Mortality in a cohort of patients with COPD	Mean not reported IQR 8.9 ppb 24-h avg	PM_{10}, O_3, CO	0-2	Conditional logistic (case-crossover)	7.8% (−2.0, 18.6) with PM_{10}: 3.9% (−12.0, 22.5)
Sunyer et al. (2002) Barcelona, Spain 1986-1995	All cause, respiratory, and cardiovascular mortality in a cohort of patients with severe asthma	1-h max: median 47 ppb; 24-h avg median 27 ppb	PM_{10}, BS, SO_2, O_3, CO, pollen	0-2	Conditional logistic (case-crossover)	Odds Ratio: Patients with 1 asthma admission: All cause: 1.10 (0.80, 1.51) Cardiovascular: 1.70 (0.96, 2.99) Patients with more than 1 asthma adm: All cause: 2.14 (1.10, 4.14) Cardiovascular: 1.53 (0.46, 5.07)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
EUROPE (cont'd)						
Díaz et al. (1999) Madrid, Spain 1990-1992	All cause; respiratory; cardiovascular	24-h avg Levels not reported.	TSP, O_3, SO_2, CO	1, 4, 10	Autoregressive linear. Time-series study.	Only significant risk estimates were shown. For NO_2, only respiratory mortality was significantly ($p < 0.05$) associated with an excess percent risk 8.5%.
Latin America						
Borja-Aburto et al. (1997) Mexico City 1990-1992	All cause; respiratory; cardiovascular; all ages; age <5 yrs; age >65 yrs	1-h max O_3: Median 155 ppb 8-h max O_3: Median 94 ppb 10-h avg O_3 (8 a.m.-6 p.m.): Median 87 ppb 24-h avg O_3: Median 54 ppb	TSP, SO_2, CO; two-pollutant models	0, 1, 2	Poisson iteratively weighted and filtered least-squares method. Time-series study.	1-h max O_3 (per 100 ppb): All ages:
Borja-Aburto et al. (1998) SW Mexico City 1993-1995	All cause; respiratory; cardiovascular; other; all ages; age >65 yrs	37.7 ppb, 24-h avg	$PM_{2.5}$, O_3, SO_2; two-pollutant models	0, 1, 2, 3, 4, 5, and multiday avg	Poisson GAM with default convergence criteria (only one smoother). Time-series study.	Lag 1-5: All cause: 2.3% (−1.0, 5.6); Cardiovascular: 2.8% (−3.2, 9.2); Respiratory: 4.7% (−5.1, 15.5).
Loomis et al. (1999) Mexico City 1993-1995	Infant mortality	24-h avg 38 ppb	$PM_{2.5}$, O_3	0, 1, 2, 3, 4, 5, 3-5	Poisson GAM with default convergence criteria. Time-series study.	Lag 3-5: 11.4% (2.2, 21.4); with $PM_{2.5}$: 2.9% (−10.2, 17.8)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
LATIN AMERICA (cont'd)						
Gouveia and Fletcher (2000b) São Paulo, Brazil 1991-1993	All ages (all cause); age <5 yrs (all cause, respiratory, pneumonia); age 65+ yrs (all cause, respiratory, cardiovascular)	1-h max: 84 ppb	PM_{10}, O_3, SO_2, CO	0, 1, 2	Poisson GLM. Time-series study.	All ages: All cause: Lag 0: −0.1% (−0.7, 0.4) Age 65+: All cause: Lag 1: 0.4% (−0.2, 1.1) Respiratory: Lag 2: 1.0% (−0.6, 2.5) Cardiovascular: Lag 1: 0.5% (−0.4, 1.3)
Pereira et al. (1998) São Paulo, Brazil 1991-1992	Intrauterine mortality	24-h avg 82 ppb	PM_{10}, O_3, SO_2, CO	0-4	Poisson GLM. Time-series study.	Single-pollutant model: 5.1% (2.8, 7.5); With other pollutants: 4.7% (1.6, 7.9)
Saldiva et al. (1994) São Paulo, Brazil 1990-1991	Respiratory; age <5 yrs	24-h avg NO_x 127 ppb	PM_{10}, O_3, SO_2, CO; multipollutant models	0-2	OLS of raw or transformed data. Time-series study.	NO_x slope estimate: 0.007197 deaths/day/ppb (SE 0.003214), p = 0.025
Saldiva et al. (1995) São Paulo, Brazil 1990-1991	All cause; age 65+ yrs	24-h avg NO_x 127 ppb	PM_{10}, O_3, SO_2, CO; two-pollutant models	0-1	OLS; Poisson with GEE. Time-series study.	NO_x slope estimate: 0.0341 deaths/day/ppb (SE 0.0105)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
LATIN AMERICA (cont'd)						
Cifuentes et al. (2000) Santiago, Chile 1988-1966	All cause	8-h avg 41 ppb	$PM_{2.5}$, PM_{10}-2.5, CO, SO_2, O_3	0, 1, 2, 3, 4, 5, 1-2, 1-3, 1-4, 1-5	Poisson GAM with default convergence criteria; Poisson GLM. Time-series study.	GLM model, lag 1-2: Single pollutant: 1.7% (0.7, 2.7); with other pollutants: 1.5% (0.3, 2.7) (per 25ppb 8-h avg)
Ostro et al. (1996) Santiago, Chile 1989-1991	All cause	1-h max 56 ppb	PM_{10}, O_3, SO_2; two-pollutant models	1	OLS, Poisson. Time-series study.	Poisson, lag 1: −0.5% (−1.1, 0)
AUSTRALIA						
Simpson et al. (2005a,b) Brisbane, Sydney, Melbourne, and Perth, Australia 1996-1999	All cause, respiratory, and cardiovascular in all ages; cardiovascular in age 65+ yrs	1-h max ranged from 16 to 24 ppb	PM_{10}, $PM_{2.5}$, bsp (nephelometer), O_3, CO	0, 1, 2, 3, 0-1	Poisson GLM, GAM with stringent convergence criteria. Time-series study.	Lag 0-1, GAM, all-cause, single pollutant: 3.4% (1.1, 5.7); with bsp: 3.1% (0.3, 5.9); cardiovascular: 4.3% (0.9, 7.8); respiratory: 11.4% (3.5, 19.9)
Simpson et al. (2000) Brisbane, Australia 1991-1996	All cause, respiratory, and cardiovascular in all ages; cardiovascular in age 65+ yrs	24-h avg: whole yr: 12 ppb; cool season: 13 ppb; warm season 9 ppb	PM_{10}, $PM_{2.5}$, bsp, O_3, CO	0, 1, 2, 3, 0-1	Poisson, GAM with default convergence criteria. Time-series study.	All-cause (lag 1): 9.7% (4.7, 14.8); respiratory: 18.8% (1.2, 39.6)
Morgan et al. (1998b) Sydney, Australia 1989-1993	All cause; respiratory; cardiovascular	24-h avg 13 ppb; 1-h max 26 ppb	bsp, O_3	0-1	Poisson with GEE. Time-series study.	Lag 0-1, single pollutant, all-cause: 3.0% (0.1, 6.0); cardiovascular: 2.2% (−1.7, 6.4); respiratory: 8.6% (−0.4, 18.4)
Simpson et al. (1997) Brisbane, Australia 1987-1993	All cause; respiratory; cardiovascular	24-h avg 14 ppb; 1-h max 28 ppb	PM_{10}, bsp, O_3, SO_2, CO	0	Autoregressive Poisson with GEE. Time-series study.	Lag 0-1, single pollutant, all-cause, all-yr: −1.0% (−5.2, 3.4); summer: −3.6% (-11.2, 4.7); winter: 1.2% (−4.0, 6.9)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
ASIA						
Kim et al. (2004b) Seoul, Korea 1995-1999	All cause	24-h avg 33 ppb.	PM_{10}, O_3, SO_2, CO; two-pollutant models	1	Poisson GAM with stringent convergence criteria (linear model); GLM with cubic natural spline; GLM with B-mode spline (threshold model). Time-series study.	Risk estimates for NO_2 not reported.
Lee et al. (1999) Seoul and Ulsan, Korea 1991-1995	All cause	1-h max O_3: Seoul: 32.4 ppb 10th %-90th % 14-55 Ulsan: 26.0 ppb 10th %-90th % 16-39	TSP, SO_2	0	Poisson with GEE. Time-series study.	1-h max O_3 (per 50 ppb): Seoul: 1.5% (0.5, 2.5) Ulsan: 2.0% (−11.1, 17.0)
Lee and Schwartz (1999) Seoul, Korea 1991-1995	All cause	1-h max O_3: Seoul: 32.4 ppb 10th %-90th % 14-55	TSP, SO_2	0	Conditional logistic regression. Case crossover with bidirectional control sampling.	1-h max O_3 (per 50 ppb): Two controls, plus and minus 1 wk: 1.5% (−1.2, 4.2) Four controls, plus and minus 2 wks: 2.3% (−0.1, 4.8)
Kwon et al. (2001) Seoul, Korea 1994-1998	Mortality in a cohort of patients with congestive heart failure	24-h avg 32 ppb	PM_{10}, O_3, SO_2, CO	0	Poisson GAM with default convergence criteria; case-crossover analysis using conditional logistic regression.	Odds ratio in general population: 1.1% (−0.3, 2.5) Congestive heart failure cohort: 15.8% (1.8, 31.7)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
ASIA (cont'd)						
Ha et al. (2003) Seoul, Korea 1995-1999	All cause; respiratory; postneonatal (1 mo to 1 yr); age 2-64 yrs; age 65+	24-h avg 33 ppb	PM_{10}, O_3, SO_2, CO	0	Poisson GAM with default convergence criteria. Time-series study.	All cause for postneonates: 0.8% (−5.7, 7.7); age 65+: 3.8% (3.7, 3.9)
Hong et al. (2002) Seoul, Korea 1995-1998	Acute stroke mortality	24-h avg 33 ppb	PM_{10}, O_3, SO_2, CO	2	Poisson GAM with default convergence criteria. Time-series study.	4.3% (1.6, 7.0)
Tsai et al. (2003b) Kaohsiung, Taiwan 1994-2000	All cause; respiratory; cardiovascular; tropical area	24-h avg 29 ppb	PM_{10}, SO_2, O_3, CO	0-2	Conditional logistic regression. Case-crossover analysis.	Odds ratios: All cause: 0.1% (−5.9, 6.6); Respiratory: −1.0% (−22.2, 25.9); Cardiovascular: −1.8% (−14.0, 12.1)
Yang et al. (2004b) Taipei, Taiwan 1994-1998	All cause; respiratory; cardiovascular; subtropical area	24-h avg 31 ppb	PM_{10}, SO_2, O_3, CO	0-2	Conditional logistic regression. Case-crossover analysis.	Odds ratios: All cause: 0.6% (−3.9, 5.2); Respiratory: 2.5% (−13.1, 20.8); Cardiovascular: −1.1% (−9.5, 8.0)
Wong et al. (2001b) Hong Kong 1995-1997	All cause; respiratory; cardiovascular	24-h avg 25 ppb in warm season; 33 ppb in cold season	PM_{10}, O_3, SO_2; two-pollutant models	0, 1, 2	Poisson GAM with default convergence criteria. Time-series study.	All cause (lag 1): 2.6% (0.9, 4.4); Respiratory (lag 0): 6.1% (−1.8, 10.5); Cardiovascular (lag 2): 5.2% (1.8, 8.7)
Wong et al. (2002) Hong Kong 1995-1998	Respiratory; cardiovascular; COPD; pneumonia and influenza; ischemic heart dis.; cerebrovascular	24-h avg 29 ppb	PM_{10}, O_3, SO_2; two-pollutant models	0, 1, 2, 0-1, 0-2	Poisson GLM. Time-series study.	Respiratory (0-1): 5.1% (1.6, 8.7); Cardiovascular (lag 0-2): 3.1% (−0.2, 6.5)

TABLE AX6.9 (cont'd). EFFECTS OF ACUTE NO_X EXPOSURE ON MORTALITY. RISK ESTIMATES ARE STANDARDIZED FOR PER 20 PPB 24-H AVG NO_2 INCREMENT

Reference, Study Location, and Period	Outcome Measure	Mean NO_2 Levels	Copollutants Considered	Lag Structure Reported	Method/Design	Effect Estimates
ASIA (cont'd)						
Hedley et al. (2002) Hong Kong 1985-1995 Intervention Jul 1990 (switch to low sulfur-content fuel)	All cause; cardiovascular; respiratory; neoplasms and other causes; all ages; age 15-64 yrs; age 65+ yrs	Avg moly NO_2: Baseline: 29 ppb 1 yr after intervention: 25 ppb 2-5 yrs after intervention: 28 ppb	SO_2 (main pollutant of interest, 45% reduction observed 5 yrs after intervention), PM_{10}, SO_4^{2-}, NO_2	Moly avgs considered without lags	Poisson regression of moly avgs to estimate changes in the increase in deaths from warm to cool season. Annual proportional change in death rate before and after the intervention was also examined.	Declines observed in all cause (2.1%, p = 0.001), respiratory (3.9%, p = 0.001), and cardiovascular (2.0%, p = 0.020) mortality after the intervention. As NO_2 levels did not change before and after the intervention, NO_2 likely did not play a role in the decline in observed mortality.
Yang et al. (2004b) Taipei, Taiwan 1994-1998	All cause; respiratory; cardiovascular; subtropical area	24-h avg 31 ppb	PM_{10}, SO_2, O_3, CO	0-2	Conditional logistic regression. Case-crossover analysis.	Odds ratios: All cause: 0.6% (−3.9, 5.2); Respiratory: 2.5% (−13.1, 20.8); Cardiovascular: −1.1% (− 9.5, 8.0)

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