EPA/600/R-03/140R
September 2003

Integrating Ecological Risk Assessment and Economic Analysis in Watersheds:

A Conceptual Approach and Three Case Studies

National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH

DISCLAIMER

ABSTRACT

This document reports on a program of research to investigate the integration of ecological risk assessment (ERA) and economics, with an emphasis on the watershed as the scale for analysis. In 1993, the U.S. Environmental Protection Agency initiated watershed ERA (W-ERA) in five watersheds to evaluate the feasibility and utility of this approach. In 1999, economic case studies were funded in conjunction with three of those W-ERAs: the Big Darby Creek watershed in central Ohio; the Clinch Valley (Clinch and Powell River watersheds) in southwestern Virginia and northeastern Tennessee; and the central Platte River floodplain in Nebraska. The ecological settings, and the analytical approaches used, differed among the three locations, but each study introduced economists to the ERA process and required the interpretation of ecological risks in economic terms. A workshop was held in Cincinnati, OH in 2001 to review progress on those studies, to discuss environmental problems involving other watershed settings, and to discuss the ideal characteristics of a generalized approach for conducting studies of this type. Based on the workshop results, a conceptual approach for the integration of ERA and economic analysis in watersheds was developed.

The objectives of this document (by chapter) are as follows:

- describe the rationale, limitations, and contributions of the document (Chapter 1)
- create a context for understanding by a diverse, technical audience (Chapter 2)
- present a conceptual approach for integrating ERA and economics in the context of watershed management (Chapter 3)
- present and critically evaluate the methods and findings of the three watershed case studies (Chapters 4-6)
- identify research needed to improve the integration of ERA and economic analysis in watersheds (Chapter 7).

This report is unique in its focus on the problem of ERA-economic integration and the watershed management context and in its presentation of case studies. The conceptual approach is used as a basis of discussion of each case study to illustrate how its particular methodological advances and insights could be used to fullest advantage, both in the watershed studied and in future integration efforts.

Preferred citation:

U.S. Environmental Protection Agency (USEPA). (2003) Integrating ecological risk assessment and economic analysis in watersheds: A conceptual approach and three case studies. Prepared by the National Center for Environmental Assessment, Cincinnati, OH. EPA/600/R-03/140R. Available from: National Technical Information Service, Springfield, VA, PB2004-101634; and <http://www.epa.gov/ncea>.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

ASCA	Alternative-Specific Constants – Option A
ASCB	Alternative-Specific Constants – Option B
AWQC	Ambient Water Quality Criteria
BMPs	Best Management Practices
BOD	Biological Oxygen Demand
CA	Conjoint Analysis
CAFOs	Confined Animal Feeding Operations
CBA	Cost-Benefit Analysis
CEA	Cost-Effectiveness Analysis
CENR	Committee on Environment and Natural Resources
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
COD	Chemical Oxygen Demand
CSO	Combined Sewer Overflow
CV	Compensating Variation
CVM	Contingent Valuation Method
CWA	Clean Water Act
DEM	Digital Elevation Models
DO	Dissolved Oxygen
DOI	Department of Interior
DPSIR	Driving forces, Pressures, State, Impacts, Response
DWR	Department of Water Resources
EIA	Economic Impact Analysis
EMAP	Environmental Monitoring and Assessment Program

EPT	Ephemeroptera, Plecoptera, and Tricloptera index
ERA	Ecological Risk Assessment
ESA	Endangered Species Act
FERC	Federal Energy Regulatory Commission
GIS	Geographic Information Systems
IBI	Index of Biotic Integrity
ICI	Invertebrate Community Index
KAF	Knowledge Adjustment Factor
KL	Knowledge Index
KL_i	Knowledge Level
MIwb	Modified Index of Well-Being
MRS	Marginal Rate of Substitution
NEPA	National Environmental Policy Act
NOAA	National Oceanic and Atmospheric Administration
NPS	Nonpoint Source
NRC	National Research Council
NRCS	Natural Resource Conservation Service
NRDA	Natural Resource Damage Assessment
OECD	Organization for Economic Cooperation and Development
OEPA	Ohio Environmental Protection Agency
POTWs	Publicly-Owned Treatment Works
PRWCMT	Platte River Whooping Crane Maintenance Trust
QHEI	Qualitative Habitat Evaluation Index
RUM	Random Utility Model

SAB	Science Advisory Board
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TNC	The Nature Conservancy
TP	Total Phosphorus
TVA	Tennessee Valley Authority
UAA	Use Attainability Analysis
UN-L	University of Nebraska-Lincoln
USACE	U.S. Army Corps of Engineers
USEPA	U.S. Environmental Protection Agency
USFS	U.S. Forest Service
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
UT-K	University of Tennessee-Knoxville
W-ERA	Watershed Ecological Risk Assessment
WQS	Water Quality Standards
WTA	Willingness to Accept
WTP	Willingness to Pay

PREFACE

A national goal of the Clean Water Act is to achieve water quality that provides for the protection and propagation of fish, shellfish, and wildlife, wherever attainable. To ensure a sound scientific basis for the protection of aquatic and other ecosystems and the diversity of species they support, the USEPA published a *Framework for Ecological Risk Assessment* in 1992 and *Guidelines for Ecological Risk Assessment* in 1998. Since the early 1990s, the USEPA has also urged the use of a "watershed approach" to aquatic ecosystem protection, which views the geographic area encompassed by a watershed as the basis for monitoring, assessment and the formation of management partnerships and action plans. The watershed is also the usual basis for establishing total maximum daily loads (TMDLs) for impaired waters.

Under Executive Order 12866 and the Unfunded Mandates Reform Act, the USEPA is required to document the costs and benefits of its major regulatory actions. To guide those efforts it published, in 2000, the *Guidelines for Preparing Economic Analyses.* Additional guidance for determining the economic benefits of ecosystem protection was provided in the 2002 *Framework for the Economic Assessment of Ecological Benefits*. More information is needed, however, about the application of economic methods to local ecological protection efforts, such as at the level of the watershed. Watersheds are varied settings in which the ecological resources, stakeholder concerns, management partnerships and decision-making arrangements tend to be unique, and flexible approaches to analysis and problem-solving are required. Furthermore, while advances continue to occur in the methods of ecological risk assessment and economics, the integration of these sciences remains problematic.

This technical report presents the results of USEPA-sponsored ecological and economic research conducted in three locations: the Big Darby Creek watershed of Ohio, the upper Clinch and Powell River watersheds of Virginia and Tennessee and the central reach of the Platte River in Nebraska. The watershed management problems that were addressed and the study techniques used differed from case to case, and they achieved varying degrees of success. The information gained from these experiences has enabled the development of a generalized conceptual approach for the integration of ecological risk assessment and economic analysis in watershed management, which this report also presents.

This report will be useful to technical audiences interested in the science and practice of watershed management and in the scientific and practical problems that underlie the integration of ecology and economics. The conceptual approach that it presents provides useful insights for the future design of integrated watershed assessments.

AUTHORS, CONTRIBUTERS, AND REVIEWERS

The National Center for Environmental Assessment (NCEA), within U.S. EPA's Office of Research and Development, was responsible for preparing this document. Randall J. F. Bruins and Matthew T. Heberling (NCEA) were the document editors.

CHAPTER AUTHORS

Chapter 1. **Randall J. F. Bruins and Matthew T. Heberling**
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 2. **Randall J. F. Bruins and Matthew T. Heberling**
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 3. **Randall J. F. Bruins and Matthew T. Heberling**
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 4. **O. Homer Erekson**
Bloch School of Business & Public Administration
University of Missouri, Kansas City, MO

Orie L. Loucks, Steven R. Elliott, and Donna S. McCollum
Departments of Economics and Zoology
Miami University, Oxford, OH

Marc Smith
Ohio Environmental Protection Agency, Columbus, OH

Randall J. F. Bruins
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 5. **Steven Stewart**
Department of Hydrology & Water Resources
The University of Arizona, Tucson, AZ

James A. Kahn
Environmental Studies Program, Williams School of Commerce
Washington and Lee University, Lexington, VA

Amy Wolfe and Robert V. O'Neill
Environmental Sciences Division
Oak Ridge National Laboratory, Oak Ridge, TN

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

Victor B. Serveiss
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Washington, DC

Randall J. F. Bruins and Matthew T. Heberling
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 6. **Raymond Supalla, Bettina Klaus, and John Allen**
Department of Agricultural Economics
University of Nebraska, Lincoln, NE

Dennis E. Jelinski
Departments of Biology and Geography
Queens University, Kingston, Ontario

Osei Yeboah
Department of Agricultural Economics
Auburn University, Auburn, AL

Victor B. Serveiss
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Washington, DC

Randall J. F. Bruins
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

Chapter 7. **Randall J. F. Bruins**
National Center for Environmental Assessment
U.S. Environmental Protection Agency, Cincinnati, OH

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

EXTERNAL REVIEWERS

Darrell Bosch, Ph.D.
Department of Agricultural and Applied Economics
Virginia Tech
Blacksburg, VA

Robert Costanza, Ph.D.
Gund Institute for Ecological Economics
University of Vermont
Burlington, VT

Peter deFur, Ph.D.
Environmental Stewardship Concepts
Richmond, VA

INTERNAL REVIEWERS

Chapter 2.

Anne Grambsch
Office of Research and Development
National Center for Environmental Assessment

Brian Heninger
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Sabrina Ise-Lovell
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Christopher Miller
Office of Water
Office of Science and Technology

Mark L. Morris
Office of Water
Office of Science and Technology

Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

William O'Neil
Office of Policy, Economics and Innovation
National Center for Environmental Economics

John Powers
Office of Water
Office of the Assistant Administrator

Keith Sargent
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Anne Sergeant
Office of Research and Development
National Center for Environmental Assessment

Victor Serveiss
Office of Research and Development
National Center for Environmental Assessment

Glenn Suter II
Office of Research and Development
National Center for Environmental Assessment

William Wheeler
Office of Research and Development
National Center for Environmental Research

Chapter 3. Wayne Munns
Office of Research and Development
National Health and Environmental Effects Research Laboratory

Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

William O'Neil
Office of Policy, Economics and Innovation
National Center for Environmental Economics

John Powers
Office of Water
Office of the Assistant Administrator

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

Keith Sargent
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Anne Sergeant
Office of Research and Development
National Center for Environmental Assessment

Victor Serveiss
Office of Research and Development
National Center for Environmental Assessment

Chapter 4. Brian Heninger
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Matt Massey
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Lester Yuan
Office of Research and Development
National Center for Environmental Assessment

Chapter 5. Susan Herrod-Julius
Office of Research and Development
National Center for Environmental Assessment

Matt Massey
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Chapter 6. Sabrina Ise-Lovell
Office of Policy, Economics and Innovation
National Center for Environmental Economics

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Catriona Rogers
Office of Research and Development
National Center for Environmental Assessment

Chapter 7. Stephen Newbold
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Keith Sargent
Office of Policy, Economics and Innovation
National Center for Environmental Economics

Victor Serveiss
Office of Research and Development
National Center for Environmental Assessment

CLEARANCE REVIEWERS

Glenn Suter II
Office of Research and Development
National Center for Environmental Assessment

Michael Slimak
Office of Research and Development
National Center for Environmental Assessment

ACKNOWLEDGMENTS

The editors wish to acknowledge Bette Zwayer, Pat Daunt, Patricia L. Wilder, Dan Heing, Teresa Shannon and Lana Wood for their assistance in the preparation of this document, and Ruth Durham, Donna Tucker and David Bottimore for management of document reviews. We acknowledge Glenn Suter II, Chris Cubbison, Mike Troyer and Haynes Goddard for participation in the review of grant proposals that formed the core of this research effort, and

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

Barbara Cook for invaluable assistance in grant management. Mike Troyer prepared maps appearing in several chapters. We also acknowledge the important work of Suzanne Marcy, and members of the USEPA Risk Assessment Forum, who initiated the watershed ecological risk assessments that provided a basis for this research, and Victor Serveiss, who later assumed leadership of the watershed ecological risk assessment effort. Finally, we acknowledge Jackie Little and Nancy Keene of TN and Associates for their assistance in the organization of a workshop held in 2001 in Cincinnati, OH, and the attendees of that workshop, many of whom are further acknowledged below.

Chapter 3

The authors wish to acknowledge Glenn Suter II for many helpful discussions in the development of this chapter.

Chapter 4

The authors wish to thank the members of the Darby Partners and members of the Big Darby Creek Watershed Ecological Risk Assessment Workgroup for their participation in the development of the risk assessment on which parts of this chapter are based. We also thank attendees of a workshop held in July 2001, in Cincinnati, OH for their comments on an early draft of this work, and in particular we acknowledge John M. Gowdy, Robert V. O'Neill, Ralph Ramey, and David Szlag for their written reviews. The views expressed in this chapter are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

AUTHORS, CONTRIBUTORS, AND REVIEWERS (cont.)

Chapter 5

The authors wish to thank the members of the Clinch and Powell Watershed Ecological Risk Assessment Workgroup for their participation in developing the USEPA assessment report, upon which this manuscript is based. Dennis Yankee and Jeff White provided GIS support and database management. We also thank attendees of a workshop held in July 2001, in Cincinnati, OH for their comments on an early draft of this work, and in particular we acknowledge Leonard Shabman, Charles Menzie, Glenn Skinner and James E. Smith for their written reviews. The views expressed in this chapter are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

Chapter 6

The authors wish to thank the members of the Middle Platte Watershed Ecological Risk Assessment Workgroup for their effort in performing activities upon which this report is based, and Nancy Pritchett for technical support. We also thank attendees of a workshop held in July 2001, in Cincinnati, OH for their comments on an early draft of this work, and in particular we acknowledge Glenn Suter II, Haynes Goddard and Ann Bleed for their written reviews. The views expressed in this chapter are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

EXECUTIVE SUMMARY

1. INTRODUCTION

Aquatic ecosystems provide many services to human society, including the supply of water, food and energy, the treatment of wastes, opportunities for recreation, and the provision of habitat for many valued species. However, by altering stream corridors, changing patterns of flow, introducing nonindigenous species, and releasing pollutants into these ecosystems, society has diminished their ability to continue providing these services. Because aquatic ecosystems have complex interactions with their surrounding landscapes, efforts to better manage and to restore these systems often focus on watersheds as basic units for analysis.

This document is concerned with two types of analysis that are both important for aquatic ecosystem management: ecological risk assessment (ERA) and economic analysis. Both have been recognized as necessary, but they have been kept largely separate in practice, and this separation can hamper management efforts.

Recommended procedures for carrying out ERA have been published by the U.S. Environmental Protection Agency (USEPA) and are widely used for regulation and management. ERA carried out at the spatial scale of the watershed is termed watershed ERA (W-ERA). Watershed management choices involve complex and uncertain trade-offs of current and future financial and ecological resources. Economics offers analytic frameworks for evaluating the trade-offs involved in choices made by individuals, firms or society. However, the integration of W-ERA and economic analysis entails theoretical, technical and procedural challenges *(Section 1.1)*.

This document reports on a program of research to investigate the integration of ERA and economics, with an emphasis on the watershed as the scale for analysis. In 1993, USEPA initiated W-ERA in five watersheds to evaluate the feasibility and utility of this approach. In 1999, economic case studies were funded in conjunction with three of those W-ERAs: the Big Darby Creek watershed in central Ohio; the Clinch Valley (Clinch and Powell River watersheds) in southwestern Virginia and northeastern Tennessee; and the central Platte River floodplain in Nebraska. The ecological settings, and the analytical approaches used, differed among the three locations, but each study introduced economists to the ERA process and required the interpretation of ecological risks in economic terms *(Section 1.2)*.

The goal of the research reported in this document was to enhance the management of aquatic ecosystems by piloting the integration of ERA and economic analysis in watersheds. This document is intended for technically educated readers with an interest in improving environmental management, including academic, government, or private researchers, and local, state, or federal environmental decision-makers. The objectives of this document (by chapter) are as follows *(Section 1.3)*:

- create a context for understanding by a diverse, technical audience (Chapter 2)
- present a conceptual approach for integrating ERA and economics in the context of watershed management (Chapter 3)
- present and critically evaluate the methods and findings of the three watershed case studies (Chapters 4-6)
- identify research needed to improve the integration of ERA and economic analysis in watersheds (Chapter 7).

The topics discussed in this document overlap with the topics of three USEPA guidance documents, the *Guidelines for Ecological Risk Assessment*, the *Guidelines for Preparing*

Economic Analyses and the *Framework for Economic Assessment of Ecological Benefits*. This report is unique in its focus on the problem of ERA-economic integration and the watershed management context and in its presentation of case studies. This research report should not be construed as guidance, and it does not replace any of those guidance documents *(Section 1.4)*.

Some limitations of this document should be recognized. First, while the case studies provide insights into the problem of ERA-economic integration, these studies themselves were not integrated in any ideal sense, since the ERA and economic components were carried out separately. Second, the problem of integrating ERA and economic analysis for environmental management in general has many facets, not all of which can be addressed in the watershed context. Therefore, care should be taken in extending the findings of this document beyond that context (*Section 1.5)*.

Notwithstanding these limitations, this document makes several unique contributions for environmental management. First, it helps risk assessors better understand how ERA procedures can be integrated with economic analysis. Second, the risk assessment perspective employed in this document also poses interesting challenges for the economist, since translating ecological risks into terms amenable to economic analysis is difficult. Third, it enables a comparison of three different approaches for ERA-economic integration. Finally, this document introduces, in Chapter 3, a new conceptual approach for integrating ERA and economic analysis in the context of watershed management *(Section 1.6)*.

2. BACKGROUND: ECOLOGICAL RISK ASSESSMENT AND ECONOMIC ANALYSIS IN WATERSHEDS AND THE NEED FOR INTEGRATION

This section provides an introduction to basic terms and concepts in ERA and economic analysis and to some of their applications to watershed management. ERA is a scientifically-based process for framing and analyzing the nature, probability and uncertainty of adverse effects from human-caused threats to ecological resources. Procedures described in USEPA's *Guidelines for Ecological Risk Assessment* include four primary phases: planning, problem formulation, analysis and risk characterization.

The **planning** process is a dialogue between risk assessors and risk managers and, where appropriate, interested and affected parties (stakeholders). The dialogue clarifies the context of the environmental decision facing officials or the public, the ecosystem management goals and objectives (including the identification of what characteristics are valued), and the information needs that the assessment should address. **Problem formulation** is a process of generating preliminary hypotheses about how human activities may cause ecological effects. It requires the identification of assessment endpoints (ecological entities that reflect the valued characteristics), the development of one or more conceptual models (such as box-and-arrow diagrams of how human activities may generate stressors, leading to effects on the endpoints), and the development of an analysis plan. **Analysis** characterizes exposure and effects. Exposure analysis describes sources of stressors, stressor transport and distribution, and the extent of contact or co-occurrence between stressors and affected organisms. Effects analysis determines what effects are thought to be elicited by a stressor, then examines the quantitative relationship between the stressor and the response, the plausibility that the stressor may cause the response (causality), and the links between particular measures of effect and the assessment endpoints. **Risk characterization** unites information about exposure and effects, in order to first estimate and then describe the risks of adverse effects of stressors. Risk characterization also describes the adequacy of data, the strength of all available lines of evidence, and the uncertainties *(Section 2.1.1)*.

Critics of the uses of ERA in decision-making have argued that assessments tend to rely too heavily on limited data, to oversimplify ecological complexities and to underestimate the likelihood of unexpected outcomes. They also have argued that assessors may be biased. Most of these criticisms are addressed, however, if assessments establish an effective planning dialogue, formulate problems appropriately, and carefully evaluate different lines of evidence, as called for in the USEPA *Guidelines (Section 2.1.2)*.

Watersheds have been used for over a century as a basis for the study and management of water resources, and since the early 1990s USEPA has urged the use of a "watershed approach" for the study and management of water quality problems. Conducting ERA on a watershed scale makes sense whenever problems exist that are not addressed simply by the establishment and monitoring of water quality standards (WQS). Examples include the presence of unusual or rare habitats or species with atypical requirements; effects caused by multiple sources or stressors; and effects due to stressors such as modification of flow or habitat for which WQS have not been established or effects of unknown cause *(Section 2.1.3)*.

Welfare economics is the study of agents (individuals, firms) making choices; it assumes that they are trying to maximize their well-being (i.e., their welfare, also termed utility). In an ideal market, agents' decisions would lead to an efficient outcome, or one in which all mutually beneficial trades have been made. In real situations, however, characteristics of the market or of the goods and services often make trade in the marketplace inefficient. Markets often fail to allocate environmental goods and services efficiently, complicating efforts to estimate the values of different levels of environmental protection *(Section 2.2.1)*.

Therefore, economists have developed nonmarket methods for estimating economic value, or people's willingness to pay (WTP) for these goods and services. A variety of methods exist for estimating nonmarket values. They can be categorized according to how the data are generated (i.e., whether preferences are revealed or stated). Revealed preference methods infer values from data on actual market choices related to the good, such as travel to a recreational site. Stated preference methods use data generated by placing individuals in hypothetical choice settings, often by use of a questionnaire. The choice settings use descriptions of hypothetical changes to environmental amenities in order to elicit values *(Section 2.2.2)*. Two of the case studies presented in this document used stated preference techniques. Chapter 4 describes a contingent valuation method (CVM) model used to value alternative development approaches in the Big Darby Creek watershed. CVM surveys ask individuals how much they would be willing to pay for a specifically described nonmarket good. Chapter 5 uses conjoint analysis (CA) to study social tradeoffs among riparian protection policies in the Clinch Valley. CA surveys ask individuals to rank or choose their most preferred option from a set of nonmarket goods. Each of the goods is described in terms of a common set of attributes, and one of the attributes is the cost of providing the good (in order to estimate economic values).

Economic values can be incorporated into analyses to help support decisions about environmental protection. Traditionally, a complete economic analysis consists of three techniques: cost-benefit analysis (CBA), economic impact analysis and equity assessment. CBA is the process of summing all the individual values, present and future, associated with a project or policy. It provides a method to calculate whether the project or policy improves efficiency based on whether there are positive or negative net benefits *(Section 2.2.3)*. Economic impact analysis is a process to quantify a variety of economic consequences of various actions. Equity assessment allows economists to understand changes in the distribution of wealth due to a policy or project *(Section 2.2.4)*. A technique similar to CBA is cost-effective analysis, which ranks alternatives that are expected to deliver comparable levels of environmental protection from lowest to highest cost.

Game theory is a type of economic analysis that is concerned with human behavior and can examine individuals interacting within a market or in situations of market failure. It entails a theory of strategic behavior where an outcome depends on many individuals' strategies and the current conditions of the situation *(Section 2.2.5)*. Chapter 6 discusses the use of game models to inform an interstate water negotiation in the Platte River watershed of Colorado, Wyoming and Nebraska.

Ecological economics is a relatively new paradigm that has sought various ways to incorporate into economic analysis the physical and biological limitations of the ecological systems that underpin economic systems. In addition to efficiency and equity, it is also concerned with determining the scale of economic activity that ecological systems can sustainably support *(Section 2.2.6)*. Analytic approaches have included various methods that propose some biophysical commodity (e.g., land or energy) as a replacement for economic welfare, as well as approaches that link ecosystem models to more conventional, welfare-based economic models *(Section 2.2.7)*.

Attempts at integrating ERA and economics under terms of the Clean Water Act (CWA) have been limited. ERA procedures have been used to determine what CWA measures are protective and whether they are physically attainable, whereas economic analyses have been used primarily to determine what is cost-effective and financially attainable. Under the CWA, states, tribes, and territories with approved WQS programs must establish designated uses or goals for their water bodies. Scientifically-derived criteria are then adopted to protect the designated uses. In general, WQS are based on a level of water quality that provides for the protection of aquatic life (i.e., propagation of fish, shellfish, and wildlife) wherever it can be reasonably attained, not wherever it can be shown to provide positive net benefits. If WQS are not being met, the costs to society of attainment can be substantial, but the benefits of attainment, often harder to measure, can be large as well. Therefore, methods for better understanding the tradeoffs between the ecological and economic effects of WQS are of interest *(Section 2.3)*.

The same holds for the attainment of nonregulatory goals. ERA is useful for determining the likely ecological responses to various kinds of proposed management actions, and economic analysis is useful for interpreting those ecological changes, and other changes, in terms of human well-being – so that decisions are effective and beneficial. But the best results will be achieved only if ERA and economic analysis are integrated, rather than compartmentalized. A coherent integration approach is needed *(Section 2.4)*.

3. A CONCEPTUAL APPROACH FOR INTEGRATED WATERSHED MANAGEMENT

Several frameworks have been applied to watershed management processes, but none has addressed specifically the ERA-economic integration problem. An approach for this purpose should be tailored accordingly, since existing frameworks vary widely in scope and purpose. Some address only monitoring or assessment, and exclude decision-making, whereas others describe planning and management processes more broadly. Some frameworks are for situational use, in response to problems or opportunities, whereas others describe regular, ongoing management processes. Frameworks also differ as to the extent to which they integrate the natural and social sciences and in the roles stakeholders are expected to play *(Section 3.1)*.

Other characteristics should also be considered in design of a new approach. According to USEPA's Science Advisory Board, processes used for integrated environmental management should be transparent (clearly understandable) to all parties; flexibly applied; dynamic (interconnected and iterative); open and cooperative; informed by many different sources and disciplines; and should reflect holistic, systems thinking *(Section 3.2)*.

This document presents a new conceptual approach for the integration of ERA and economic analysis in watersheds (Figure ES-1). The approach is designed so as to recognize the unique value of ERA, to be responsive to critiques of ERA, to incorporate key attributes of economic thought, to be pluralistic in methodology, to incorporate adaptive management (i.e., a learning-by-doing approach) and to link situational and regular management processes. It borrows from USEPA's *Framework for Ecological Risk Assessment*, but modifies that approach at every stage to integrate economic analysis.

Assessment planning is analogous to "planning" in ERA, except that identification of the decision context is expanded to include determining who has the authority to make the decisions and what criteria they expect to use. **Problem formulation** is also similar to that done in ERA, except that economic as well as ecological assessment endpoints must be identified, and the relationships that are diagramed in conceptual models must include hypotheses about how the various management alternatives would affect the ecological and economic assessment endpoints. **Analysis and characterization of baseline risk** corresponds to the ERA stages of analysis and risk characterization but is limited to risks that exist now, or will occur in the future, if no new management action is taken. **Formulation of alternatives** entails the development of alternative action plans for achieving the watershed management objectives. It is required for integrated analysis, since economic analysis generally requires the evaluation of alternatives. **Consultation with the extended peer community** refers to deliberation with scientific peers as well as with stakeholders who have practical knowledge that is relevant to the situation.

Analysis and characterization of alternatives is the stage in which the management alternatives are assessed from the perspectives of ERA and economics (and possibly other disciplines such as human health risk or sociocultural assessment). Ecological risk characterization describes probabilities, magnitudes and severities of effects on ecological

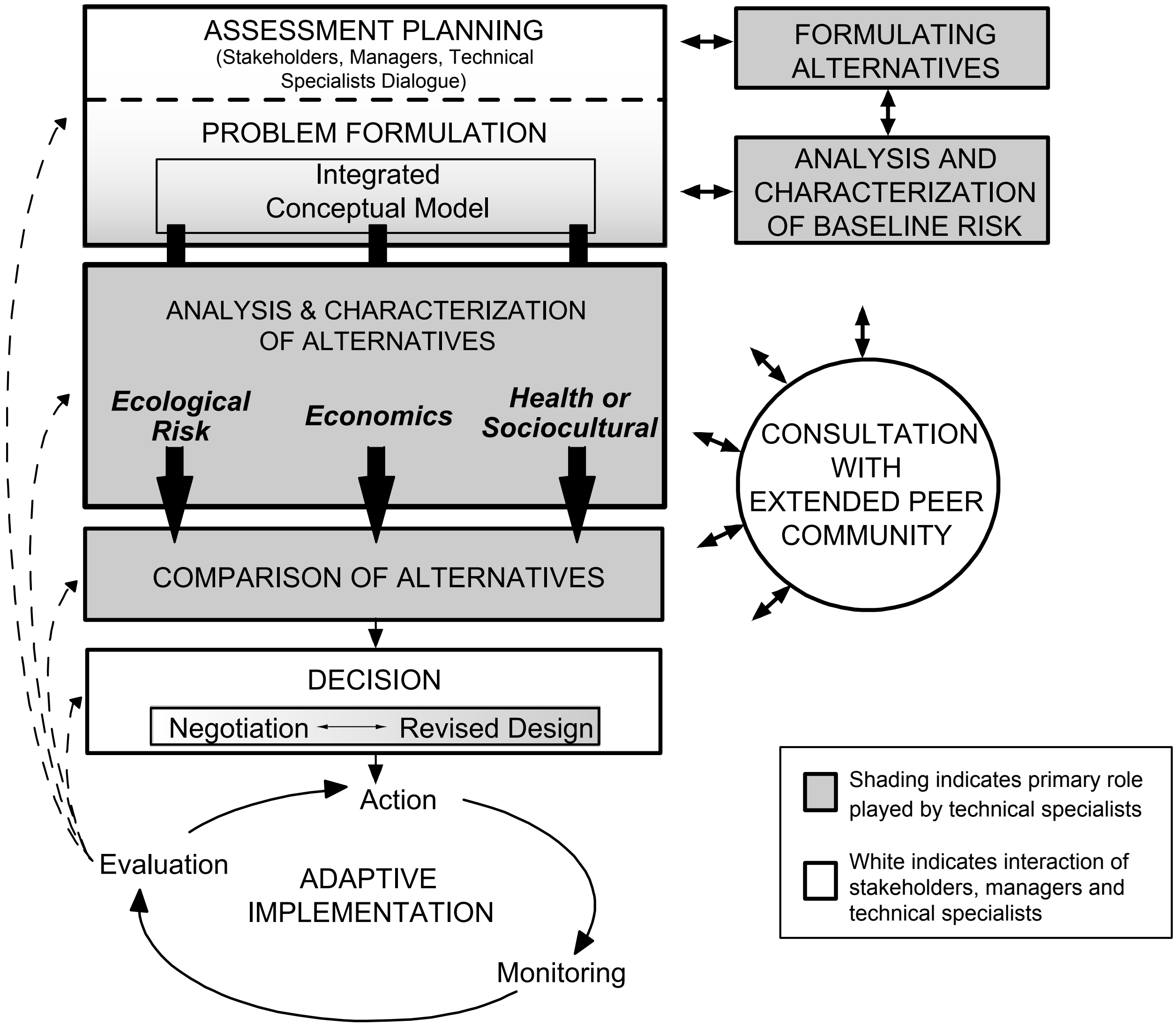

FIGURE ES-1

A conceptual approach for the integration of ecological risk assessment and economic analysis in watershed management assessment endpoints. The economic component analyzes costs and benefits associated with the management alternatives, including changes in ecosystem services. **Comparison of alternatives** is the step in which the ecological, economic and other factors, both qualitative and quantitative, are arrayed for comparison. Depending on the decision context, comparison methods could include stated preference methods, methods for assigning weights to different factors according to their importance, or methods for modeling a negotiation process. Because watershed management issues are often complex, the **decision** stage is likely to involve multiple parties and may take the form of negotiation. **Adaptive implementation**, in which management actions are monitored for effectiveness and periodically reevaluated, can help ensure that objectives are met. It can also provide a means whereby parties who are at odds can agree on an interim step that will be reevaluated after an agreed period. New information acquired during adaptive implementation may require earlier stages of assessment to be revisited. The activities of this conceptual approach are carried out only when situational needs arise, but they may be most effective when linked to regular activities such as those of the watershed management cycle used by many states *(Section 3.3)*.

The more technical steps of integration, occurring in the analysis and characterization of alternatives and in the comparison phase that follows, can employ a variety of ecological and economic analytic tools. For example, analysis and characterization could involve estimating monetary values for as many ecological and other changes as possible and using CBA to estimate the overall net benefit of each alternative. Comparison would involve examining the net benefits of the alternatives, in light of their impacts, equity effects, and any other effects that could not be quantified *(Section 3.4.1)*. In another example of an approach that could be used, ecological effects, (market based) economic effects and other effects could be quantified to the greatest extent feasible in the analysis and characterization phase, and the most important of these changes could be used in the design of a broadly-based stated preference study. Variants of this approach are used in Chapters 4-6 *(Section 3.4.2)*. Another possible approach could involve the use of linked ecological and economic models to allow ecological-economic feedbacks and optimize the design of alternatives *(Section 3.4.3)*.

4. EVALUATING DEVELOPMENT ALTERNATIVES FOR A HIGH-QUALITY STREAM THREATENED BY URBANIZATION: BIG DARBY CREEK WATERSHED

Located in central Ohio, Big Darby Creek is widely recognized for its unusual biological diversity, including many rare and endangered fish and freshwater mussel species; local efforts to protect the watershed are longstanding. However, agricultural land uses, and rapidly increasing urban development in the eastern portions of the watershed near Columbus, threaten the stream's ecological quality. The watershed was selected for W-ERA because of broad interest in protecting the Big Darby Creek and because of Ohio's large water quality database *(Section 4.1)*.

W-ERA was initiated in 1993 by USEPA, the Ohio Environmental Protection Agency and other partners. The management goal for the watershed, arrived at through planning discussions with residents, resource managers, public agencies and private organizations, was protecting and maintaining native stream communities of the Big Darby ecosystem. In the problem formulation phase, "species composition, diversity, and functional organization of the fish and macroinvertebrate communities" was chosen as the assessment endpoint. Preliminary analyses (which were expanded to encompass other areas of the Eastern Corn Belt Plains ecoregion in Ohio) showed a negative association between urban development and the functional organization of fish communities (as measured by the index of biotic integrity or IBI). Risk characterization in the watershed has not yet been completed *(Section 4.2)*.

The specific objectives of the economic case study initiated in 1999 by Miami University were as follows: (a) to estimate the quantitative or qualitative ecological and socioeconomic impacts of four land use scenarios (Preserve agriculture; Zone for low density, ranchette style; Zone for low density, cluster style and Take no action, allow high density urbanization); (b) to communicate these impacts to the public effectively, and to measure the overall economic value corresponding to each scenario based on individual willingness to pay (WTP) and (c) to better understand the particular contribution stream ecological condition makes to the value of a given scenario.

Presentations were made to three samples of respondents (residents, near-residents, and non-residents), explaining the scenarios and their likely impacts on stream ecological condition, local economic well-being and local quality of life. Using the CVM, respondents were asked their WTP to avoid the high density urban scenario given one of the other three remaining scenarios. The results suggest that the cluster-style alternative was preferred to the agriculture or ranchette alternatives. In addition, residents were willing to pay more than near-residents, and near-residents were willing to pay more than non-residents. Researchers also made a preliminary attempt to associate the WTP with a unit change in the IBI *(Section 4.3)*.

This case study demonstrated an effective use of the planning and problem formulation processes to initiate a baseline W-ERA, as well as an effective use of ecological risk information to frame a valuation question. Its value for decision-making still is limited, resulting in part from the separate conduct of the ERA and economic components. For example, the planning and problem formulation stages of W-ERA did not characterize a specific decision context. The economic study did not provide enough information to estimate the net social benefits or equity effects of the scenarios, because costs to current landholders were not estimated. To better determine the applicability of WTP measured in this study to watershed management, a renewed assessment planning process focusing on development decisions would be needed. Further work also is needed to determine the component of WTP that is specifically attributable to ecological effects *(Section 4.4)*.

5. VALUING BIODIVERSITY IN A RURAL VALLEY: CLINCH AND POWELL RIVER WATERSHED

Originating in southwestern Virginia and extending into northeastern Tennessee, the watershed of the Clinch and Powell Rivers historically contained one of the most diverse fish and mussel assemblages in North America. Most evidence suggests land uses such as mining, agriculture, urbanization and other human activities are responsible for the decline and extinction of many of these populations. This area was chosen as a subject of W-ERA because of its remaining valued aquatic resources, the wealth of information already collected, interest from many groups, and the multiple stressors present *(Section 5.1)*.

W-ERA was initiated in 1993 by USEPA, the U.S. Fish & Wildlife Service (USFWS), The Nature Conservancy and other partners. An interagency workgroup determined the management goal to be "establish[ing] and maintain[ing] the biological integrity of the Clinch/Powell watershed surface and subsurface aquatic ecosystem." The two assessment endpoints selected were: (1) reproduction and recruitment of threatened, endangered or rare native freshwater mussels and (2) reproduction and recruitment of native, threatened, endangered or rare fish species. Analyses examined various correlations between land uses, instream habitat quality, IBI and mussel diversity. The assessment found that stream reaches with high portions of riparian areas in agriculture had poor in-stream habitat and low IBI values, stream reaches close to mining activity had low IBI values, and stream reaches with many stressors present had low numbers of mussel species and low IBI values *(Section 5.2)*.

The economic analysis, initiated in 1999 by a team headed by researchers from the University of Tennessee-Knoxville, addressed the difficult task of valuing potential changes in biological diversity and other ecological services at risk in the watershed. As a focus of analysis, researchers examined hypothetical, voluntary policies to restrict agriculture in the riparian zone with compensation to farmers. Using a conjoint analysis (CA) survey, watershed respondents were asked to choose between alternative descriptions of the watershed as a function of the agricultural policy and certain other characteristics. Those characteristics dealt with recovery of aquatic life, quality of sport fishing, prevalence of song birds, effects on agricultural income, and cost per household.

The responses provide information on the quality-of-life trade-offs respondents were willing to make among various ecological and economic characteristics of this watershed. The resulting choice model provides the values respondents would place on a range of policy changes similar to those identified in the survey. This ability to estimate welfare effects over a complex set of ecosystem changes is an advantage of CA over other valuation techniques *(Section 5.3)*.

The economic study made effective use of qualitative information from the W-ERA study to design the CA survey, and the study demonstrates the flexibility of the CA method. However, because the ecological and economic effects of the policies themselves were not quantitatively characterized, these results are of limited use for policy evaluation without additional analyses. Furthermore, as in the Big Darby Creek case study, the decision context relevant to the establishment of riparian management policies (i.e., who makes these decisions and how they are made) would need to be further explored before the usefulness of this approach for management could be determined *(Section 5.4)*.

6. SEEKING SOLUTIONS FOR AN INTERSTATE CONFLICT OVER WATER AND ENDANGERED SPECIES: PLATTE RIVER WATERSHED

Nearly one-half million sandhill cranes and several million ducks and geese use the central Platte River floodplain in Nebraska during their annual migration. Several species that depend on its broad, braided channel and associated wet meadow habitats -- including the interior least tern, the piping plover and the whooping crane -- are federally listed as threatened or endangered. However, flow diversions and storage reservoirs that supply irrigation, hydropower and recreation to the region's economy are jeopardizing these habitats and species, sparking conflict among federal agencies and water users in Nebraska, Colorado and Wyoming. USFWS has determined an amount of annual flow and an acreage of restored wet meadows required for meeting species' needs; the states have negotiated lesser amounts, to be implemented on a trial basis and monitored for ten years, but since they still disagree as to who should provide those reduced amounts, action has been delayed for several years *(Section 6.1)*.

Interest in protecting these ecological resources, and willingness of several agencies and stakeholders to participate, led to the establishment in 1993 of a W-ERA workgroup. The management goal was to "protect, maintain and, where feasible, restore biodiversity and ecological processes in the central Platte River floodplain, to sustain and balance ecological resources with human uses." Nine assessment endpoints were derived from this broad goal, but analyses were completed only for grassland breeding bird diversity and abundance and sandhill crane abundance and distribution. Habitat use by wet-meadow nesting species was maximized in larger patches, suggesting that habitat fragmentation has adverse effects on these species. Use of river segments by sandhill cranes was found to be a function of channel width and the proximity of wet meadows. However, a characterization of the risks to these species, especially in relation to stream flow, was not completed *(Section 6.2)*.

An economic analysis initiated in 1999 by the University of Nebraska-Lincoln studied game theory as a means to identify policies that might help resolve the Platte River resource management conflict. Two models were constructed. Model I demonstrated a simple auctioning approach for supplying the needed water whereby the players (the three states) would have incentives to reveal their true supply costs. Model II, a multilateral bargaining model, sought to identify promising policy solutions by examining (a) different ways to provide additional water and habitat, (b) how far the parties are willing to go toward meeting the USFWS requirement and (c) how costs should be shared. Constructing the model required surveying a sample of households in the three states to correlate attitudes on these policy questions to membership in interest groups (state residency, agricultural, environmental). The survey also evaluated respondents' level of knowledge about the factors affecting these species. The survey found the greatest level of disagreement was between agricultural and environmental interests within a state, rather than among states. Policies finding widest acceptance involved adaptive (trial) implementation, minimization of impacts on agriculture, and a partial sharing of costs by environmental interests *(Section 6.3)*.

The limited interaction between risk assessors and economists in this case study resulted in a divergence of analytic objectives and perspectives. The W-ERA did not address a particular decision context, whereas the economic study developed a tool designed to inform a specific negotiation process. The W-ERA studied habitat requirements of dozens of riparian-dependent avian species while the economic analysis addressed only the needs of endangered species. Game theory models may be well-suited to the support of ongoing negotiation because they can respond quickly to changes in negotiating position and can suggest new solutions. However, the solutions will not necessarily result in protection of species unless (a) interest groups are informed about species' needs and choose to support them or (b) informational feedback from the adaptive implementation process addresses key questions and is used to update the policies *(Section 6.4)*.

7. CONCLUSIONS

The following conclusions are derived from evaluation of the case studies:

- Achieving ecological-economic integration requires a coherent strategy, such as the conceptual approach presented in Figure ES-1
- Integration requires assessment planning and problem formulation to be interdisciplinary, involving ecologists and economists (and other disciplines as needed)
- Research is needed on the development and use of integrated conceptual models, i.e., models that include economic as well as ecological endpoints and show how management alternatives are expected to affect those endpoints
- Clearly formulated management alternatives facilitate integrated analysis by giving risk assessors and economists a common basis for analyzing endpoint changes
- Careful effort is required to relate ecological endpoints to economic value, including linking these endpoints to ecosystem services and devising methods for explaining ecological measurements or indices to the public
- The appropriate tools for analysis and comparison of alternatives depend on the decision context, and since decision situations in watershed management are varied, a variety of tools are needed
- Research is needed on appropriate means of transferring the value of ecological endpoint changes from one watershed setting to others
- The role of ecological risk information in the measurement of preferences requires further research, since individuals who are surveyed may be unfamiliar with an issue and may form their preferences based on information provided in a questionnaire.

1. INTRODUCTION

1.1 THE IMPORTANCE OF INTEGRATED, WATERSHED-LEVEL ANALYSIS

Aquatic ecosystems provide many services to human society. They mediate the supply of water for drinking and other human uses; they assimilate wastes and provide food, energy, and habitat for many valued species; they offer opportunities for transportation and recreation; and they provide aesthetic values and inspiration. In taking advantage of these services, humans have stressed these ecosystems. Alteration of stream corridors, changes in patterns of flow, introduction of nonindigenous species, and pollution by toxicants, nutrients, sediments, heat, and oxygen-demanding substances have diminished aquatic ecosystems' ability to continue providing the services that society values.

As social awareness has increased, efforts have been made to better manage and reduce human impacts upon these ecosystems. In the U.S., these efforts have included increased regulation and mitigation of pollution; increased attention to the ecological impacts of water resource projects; modification of agricultural practices and subsidies; and efforts by urban, suburban and rural communities to better steward their aquatic ecological resources through monitoring, planning and collective action. Most of these efforts have been accompanied by a recognition that aquatic ecosystems have complex interactions with their surrounding landscapes. As a result, the watershed increasingly is seen as a basic unit for aquatic ecosystem analysis and management.

This document is concerned with two types of analysis that are both important for aquatic ecosystem management: ecological risk assessment (ERA) and economic analysis. Both have been recognized as necessary, and their use is provided for in law and regulation, yet because they arise from very different philosophical traditions they have tended to remain separate in both theory and practice.[1,2] This separation hampers environmental management. Analysts from the respective traditions often fail to coordinate their efforts, lack the ability to understand one another's terminology and approach, or disagree as to what is important, and they may provide decision-makers with incomplete or confusing information. Decision-makers may also assume that these analyses ought to be separate and fail to recognize the wealth of insight that their effective integration could produce.

ERA has been defined as "a process for collecting, organizing and analyzing information to estimate the likelihood of undesired effects on nonhuman organisms, populations or ecosystems."[3] Recommended procedures for carrying out ERA have been published by the U.S. Environmental Protection Agency (USEPA),[4] and the practice has been employed for a wide variety of ecological problems and settings. For example, a 1999 report by the Committee on Environment and Natural Resources (CENR) documented the use of ERA by five U.S. federal agencies — to regulate the uses of toxic substances and pesticides, for the control of nonindiginous species, and to remediate and determine compensation for damage caused by chemical releases.[5] The general principles of ERA also underlie many important regulatory protections for aquatic ecosystems in the U.S., such as state-issued water quality standards (WQS), but watersheds themselves are not usually the subject of ERA. However, routine management approaches, including the monitoring and enforcement of WQS, cannot address certain kinds of aquatic ecosystem impairment. Some undesired effects are caused by human-caused insults (hereafter termed "stressors") for which there are no standards; these include, for example, introduced organisms and altered habitat. Some are a complex result of multiple kinds of stressors; and in some cases the causes remain unclear without further study. Moreover, some aquatic ecosystems host unique resources (such as rare species or habitats) having special requirements that are not adequately understood. In addition, it is often unclear, without focused analysis, whether a given set of proposed actions to correct these problems will be effective. In these cases, an ERA that is carried out at the spatial scale of the watershed, here termed watershed ERA (W-ERA), may be useful.

As described in Section 2.1, W-ERA focuses on the key ecological resources and management goals for the watershed, rather than regulatory standards alone. The approach directly engages stakeholders in the determination of assessment goals and scope, identifies all relevant threats, and applies scientific methods to the identification of causes, risks and uncertainties of adverse effects. The resulting information is intended to be useful for the design of approaches for ecosystem protection or restoration, whether these measures are physical or institutional, regulatory or driven by incentives, governmental or community-based – or some combination of these.

In 1993, USEPA initiated W-ERA in five watersheds to evaluate the feasibility and usefulness of this approach (Figure 1-1).[5,6] The outcomes from some of these assessments, and their usefulness for management, have been described in the literature,[7-12] and W-ERA guidance has been made available as a web-based training unit.[13] Prior to this document, however, no information has been available on approaches for integrating economic analysis with ERA in a watershed management context.

Economists study choices made by individuals or other entities relating to the allocation of scarce resources across competing uses (see Section 2.2), and economic analysis sometimes has been used jointly with ERA in support of decisions (see CENR[5] and Chapter 3). Watershed management choices involve complex and uncertain trade-offs of current and future financial

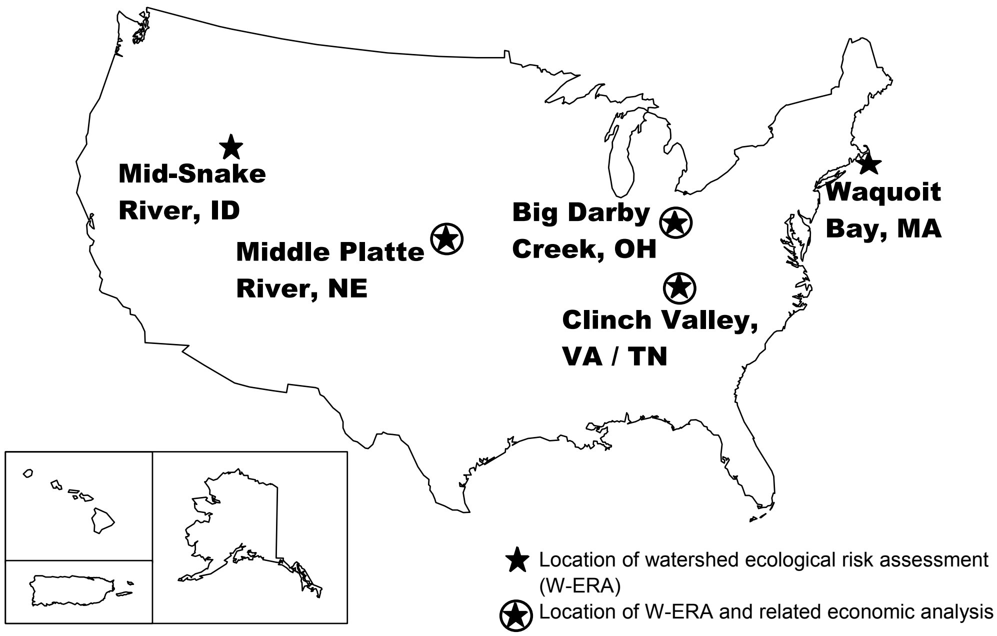

FIGURE 1-1

Locations in the USA of five watershed ecological risk assessment studies undertaken by USEPA and other partners. Comparison economic analyses were undertaken at three of the five locations as indicated.

and ecological resources. Economics offers an analytic framework for determining whether a given choice appears to provide an overall benefit to society. Depending on the approach used, economic analysis can also address impacts on affected parties, can illuminate negotiation processes, and can help evaluate the long term sustainability of outcomes. However, the integration of W-ERA and economic analysis, which is needed to realize these insights, entails theoretical, technical and procedural challenges.

1.2. GENESIS OF THIS DOCUMENT

This document reports on a program of USEPA-funded research to investigate the integration of ERA and economics, with an emphasis on the watershed as the scale for analysis. In 1998, the National Center for Environmental Assessment of USEPA's Office of Research and Development solicited applications for assistance to conduct case studies of the integration of ERA and economic analysis. Research to be funded was required to include original economic analysis conducted in collaboration with an ongoing ERA, to reflect the state of the science of ERA and economics, and to be relevant to decision-making with respect to the problem being assessed. In 1999, following peer review of proposals, economic case studies were funded in conjunction with three of the five aforementioned W-ERAs (Figure 1-1, Table 1-1).

The resulting case studies were quite different from one another. The ecological settings and resources of concern differed among the three locations. The degree of progress made by each W-ERA team prior to the economic study varied as well, and the methodological lenses brought to these problems by the respective economic teams also varied considerably. But the commonalities between these three studies were also considerable in that each involved the watershed scale, each introduced economists to the ERA process, and each included the

TABLE 1-1

Case studies of the integration of watershed ecological risk assessment and economic analysis, funded by USEPA in 1999

Study Area	Project Title	Principal Investigators
Big Darby Creek watershed, Ohio	"Determining biodiversity values in a place-based ecological risk assessment"	O. Homer Erekson and Orie L. Loucks Miami University, Oxford, Ohio
Upper Clinch Valley, Virginia and Tennessee	"A trade-off weighted index approach to integrating economics and ecological risk assessment"	James Kahn and Steven Stewart University of Tennessee-Knoxville
Central Platte River floodplain, Nebraska	"A strategic decision modeling approach to management of the middle Platte ecosystem"	Raymond Supalla University of Nebraska-Lincoln

challenging task of interpreting ecological risks in economic terms, in a manner that would be meaningful to decision-makers.

Building on those commonalities, a workshop was held in Cincinnati, OH in 2001 to review progress on those studies, to discuss environmental problems involving other watershed settings, and to discuss the ideal characteristics of a generalized approach for conducting studies of this type. Based on the workshop results, a conceptual approach for the integration of ERA and economic analysis in watersheds was developed.

1.3 OBJECTIVES AND ORGANIZATION

The goal of the research reported in this document was to enhance the management of aquatic ecosystems by piloting the integration of ERA and economic analysis in watersheds. This document is intended for technically educated readers with an interest in improving environmental management, including academic, government or private researchers, and local, state or federal environmental decision-makers. This section describes the specific objectives of this document (by document chapter).

1.3.1 Create a context for understanding by a diverse, technical audience (Chapter 2)

Because of the differences in approach between ERA and economic analysis, most readers will not be familiar with the methods and terminology of both. Therefore, Chapter 2 provides background information on ERA (Section 2.1) and economic analysis (Section 2.2) and their applications to watersheds, with special reference to their relationship to WQS programs (Section 2.3). Readers already familiar with any of these topics may skip the corresponding sections of Chapter 2.

1.3.2 Present a conceptual approach for integrating ERA and economics in the context of watershed management (Chapter 3)

Chapter 3 presents a conceptual approach for the integration of ERA and economic analysis in watershed management. This approach serves as a point of reference for critical discussion of the three case studies, and it is intended to be useful for the design of future studies that inform watershed decision-making. The chapter first reviews a variety of procedural approaches that have been applied to the study and management of watershed problems. It then identifies the main considerations that should guide the design of a conceptual approach and describes such an approach.

1.3.3 Present and critically evaluate the methods and findings of three case studies (Chapters 4-6)

Chapters 4 - 6 present detailed discussion of work done in each of the three watersheds (Table 1-1). The organization of these chapters reflects the development of these studies. In each case, a W-ERA was initiated first, by USEPA and other governmental and nongovernmental partners. The complementary economic study was initiated later, through a research grant to an educational institution. Therefore, after an initial section describing the watershed setting, the second section of each case study chapter describes the methods and findings of the W-ERA. The third section is devoted to the economic study, and a fourth section critically analyzes the success of the integration and the usefulness of results for improving management decisions.

1.3.4 Identify research needed to improve the integration of ERA and economic analysis in watershed (Chapter 7)

This final chapter reexamines the commonalities of these studies to draw general conclusions with respect to the integration problem, and it identifies areas for further research.

1.4 RELATIONSHIP TO EXISTING USEPA GUIDANCE DOCUMENTS

1.4.1 USEPA *Guidelines for Ecological Risk Assessment*

USEPA published a *Framework for Ecological Risk Assessment* in 1992,[14] and *Guidelines* in 1998.[4] These documents provide the basis for ERA as currently practiced in USEPA and many other organizations. A further guidance document that provides detail on the development of management objectives in the ERA planning process is currently in draft form.[15] These methods, summarized in Section 2.1, formed the basis for the W-ERA studies described in this document. The conceptual approach presented in Chapter 3 is based on those methods, but shows how they may be modified and extended to enable the integration of ERA and economic analysis in a watershed management context.

1.4.2 USEPA *Guidelines for Preparing Economic Analyses*

These *Guidelines*[16] describe how USEPA conducts economic analyses of its environmental policies and programs, as may be required for their justification under Federal statute or Executive Order. They present methods for deriving monetary estimates of the costs and benefits of those policies or programs. By contrast, the present document addresses watershed management processes, which are location- and context-specific and can encompass a wide variety of decision-making approaches, from statutory to ad hoc, taking place within or outside of Federal agencies and involving single- or multi-party decisions. These decisions can be informed by various economic methods, not all of which develop monetary estimates. Therefore, the present document serves a different purpose and audience. While it includes some methods that monetize ecological costs and benefits, it is not limited to them.

1.4.3 USEPA *Framework for Economic Assessment of Ecological Benefits*

This recently-developed *Framework*[17] deals specifically with the problems of integrating ERA and economic analysis, and in this context it is a valuable companion reference to the present work. Like the present document, it provides information about ERA and economic analysis to a multidisciplinary audience, and it discusses integration approaches. Like the *Guidelines for Preparing Economic Analyses*, however, it is limited to the development of monetary estimates as needed to support policy or regulation. Unlike the present work, it does not address place-based management processes, and it does not evaluate case examples.

1.5 LIMITATIONS

1.5.1 Lack of complete integration

Although the subject of this document is the integration of ERA and economic analysis, the case studies that it presents are not integrated in a complete or ideal sense. On one hand, the efforts invested by USEPA and its partners to conduct W-ERA in a set of U.S. watersheds offered a unique opportunity to sponsor complementary research in economic analysis. The assistance award criteria ensured that the funded economic research would focus on key elements of a W-ERA. Yet, as explained above, the economic studies were initiated several years later than the W-ERA studies. There was collaboration between members of the W-ERA and economic teams in each watershed, but because of the later starting point and separate funding mechanism of the economic research, the teams were not unified. Further, the conceptual approach for integration described in Chapter 3 was designed as an outcome of this research and was not available at the outset. As a result, the initial planning and problem formulation work conducted in each watershed did not include economists or consider their needs. While the economic research teams had the benefit of groundwork laid by the W-ERA effort, they sometimes perceived watershed needs and goals differently, and some of these differences are evident in this report. Finally, because of the difficulties involved in funding, coordinating and completing large, multi-participant studies, the W-ERA studies themselves were not all completed during the time frame of economic study, and in this manner as well the economists did not obtain the full benefit of interdisciplinary collaboration. Therefore, these case studies should be seen as providing a unique set of insights into the ERA-economic integration problem but not as exemplars of such integration.

1.5.2 Specificity to a watershed context

The impetus for this research is the protection of aquatic ecological resources, which often requires analysis at the level of the watershed. The problem of integrating ERA and economic analysis for environmental management in general has many facets, not all of which can be addressed in the watershed context. W-ERA tends to be resource-based; that is, it identifies the ecological resources of concern in a given place and identifies the risks to those resources. Economic analysis that is done in conjunction with W-ERA must address those risks, and the particulars of the local decision context. By contrast, policies or regulations promulgated at the national or state level (e.g., WQS, effluent guidelines) tend to address stressors or categories of polluting activities occurring over a broad area, and therefore their risk and economic assessments may have a different character. Furthermore, some integrated assessments are for the purpose of setting priorities among different kinds of environmental problems across different resources, stressors or media. Therefore, while the findings of this document shed light on the overall integration problem, they should not be considered generally applicable.

1.6 UNIQUE CONTRIBUTIONS

Notwithstanding these limitations, this document makes several unique contributions for environmental management. First, it places economic analysis into a context that is familiar to risk assessors. Because it uses the specific procedures and terminology of ERA, it will help ERA practitioners better understand how those procedures can be integrated with economic analysis. The conceptual approach presented in Chapter 3 borrows heavily from USEPA's ERA *Framework*. The case studies demonstrate how risk assessment outcomes – i.e., probabilities of adverse changes in ecological assessment endpoints – figure into economic analysis, and they sensitize the reader to the difficulties that economists face in using those results. They also illustrate for risk assessors the importance of the "with-without" context that is familiar to economists. Whereas risk assessors sometimes focus only on identifying risks associated with current situations and trends, or on identifying exposure targets for reducing those risks, economists most often focus on choices between alternative actions. Therefore, economists demand a comparison of current and future risks "with and without" a given action. The economist's perspective, evident both in the conceptual approach and the case studies, prods the risk assessor to use ERA in a way that maximizes its value to decision-makers.

Second, the risk assessment perspective employed in this document also poses interesting challenges for the economist. Economists sometimes use relatively vague statements about the ecological improvements expected under a given policy to elicit the monetary amounts individuals would pay to obtain the policy. ERA, on the other hand, uses best-available data and methods to quantify the linkages between human activities, the stressors they produce, and the ensuing effects on particular ecological endpoints. The resulting statements about risk are as specific as possible about the nature and magnitude of effects expected, but they may also include description of uncertainties. Translating these statements into terms amenable to economic analysis is difficult, as these case studies illustrate, but the challenge must be accepted if these sciences are to be integrated.[18]

The document makes a further, useful contribution by allowing comparison of different integration approaches. Two case studies used surveys to estimate economic values associated with policies to protect watershed ecological resources, based on the assessment endpoints identified in the W-ERA. One of these (see Chapter 4) valued those policies explicitly, using the contingent valuation method, whereas another (see Chapter 5) did so implicitly, using conjoint analysis (Appendix 2-A compares these methods). The third case study (see Chapter 6) used economic game theory to identify policies most likely to resolve a longstanding conflict over the protection of watershed resources. These differences in approach make the overall findings of this document more robust.

Finally, this document introduces a conceptual approach for integrating ERA and economic analysis, in the context of watershed management as practiced under the Clean Water Act (see Chapter 3, and especially Figure 3-1). The approach draws its elements from existing USEPA guidance, as well as from other environmental management frameworks developed by various agencies and advisory bodies. By synthesizing these elements in a way that emulates yet expands the ERA *Framework*, which is a familiar tool in the field of environmental management, it communicates the essential principles of integration to an important audience.

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2. BACKGROUND: ECOLOGICAL RISK ASSESSMENT AND ECONOMIC ANALYSIS IN WATERSHEDS AND THE NEED FOR INTEGRATION

This document presents a conceptual approach and three case studies for the improved integration of ecological risk assessment (ERA) and economic analysis in the management of watersheds. This chapter lays necessary groundwork for the technically trained reader who may not have a background in ERA or in economic analysis. It explains the basic elements of each and their uses in watershed management, and helps the reader understand their uses in the case studies.

Readers already familiar with the U.S. Environmental Protection Agency's (USEPA's) *Guidelines for Ecological Risk Assessment* [1] can safely skip Section 2.1.1, which summarizes the steps of ERA, but should read Sections 2.1.2 and 2.1.3, on its critiques and watershed applications, respectively. Similarly, readers acquainted with environmental economics need not read Sections 2.2.1 through 2.2.4, which cover familiar theory and applications, but they may want to read Sections 2.2.5, on game theory, and 2.2.6, on ecological economics. Section 2.3 discusses applications of ERA and economics to water quality standards (WQS) programs in the U.S., and Section 2.4 offers concluding thoughts on the need for ERA-economic integration.

2.1 ECOLOGICAL RISK ASSESSMENT

This section discusses ERA and its relationship to watershed management. The goal is to provide sufficient background to make the succeeding chapters understandable to non-practitioners of ERA; it is not a comprehensive introduction to the topic. First, the origins of risk assessment and ERA in particular are briefly discussed, and the steps of ERA are presented as described in the USEPA's *Guidelines for Ecological Risk Assessment*.[1] Some criticisms of ERA are then discussed, and finally some applications of ERA to the analysis and management of environmental problems at the watershed scale are covered.

2.1.1 Framework and methods for ecological risk assessment

The U.S. Council on Environmental Quality has defined risk as "the possibility of suffering harm from a hazard" – where a hazard is "a substance or action that can cause harm" – and risk assessment as "the technical assessment of the nature and magnitude of risk."[2] The Presidential/Congressional Commission on Risk Assessment and Risk Management defined risk as "the probability of a specific outcome, generally adverse, given a particular set of conditions" and risk assessment as "an organized process … to describe and estimate the likelihood of adverse health outcomes…."[3] Risk assessment thus includes both qualitative description (i.e., the "nature" of a possible "harm") and quantitation (i.e., of its "magnitude"). "Magnitude" can apply both to the harmful effect itself (e.g., how many individuals or populations will be harmed, and to what degree) and to the possibility that the harm will occur. "Possibility" encompasses the concepts of probability (or likelihood) and uncertainty. In common usage the term "risk" often equates to likelihood, but in risk assessment a naked probability has little meaning apart from a qualitative and quantitative description of the probable harm and of the uncertainty associated with both the harm and its probability. This document uses the term "adverse effects" rather than "harm," and it uses "risk" to encompass the nature, probability and uncertainty of adverse effects.

The terms "probability" and "uncertainty" are closely related. "Uncertainty with respect to natural phenomena means that an outcome is unknown or not established and is therefore in question."[4] Uncertainty that is attributable to natural variability ("inherent uncertainty") is considered irreducible and often is described using probability distributions. Uncertainty that is due to incomplete knowledge ("knowledge uncertainty") is considered reducible given additional information.[4,5,a]

ERA is a scientifically-based process for framing and analyzing human-caused risks to ecological resources.[1,6-8] In some of its elements it follows a framework defined earlier for human health risk assessment,[9] but it differs because of special problems presented in the assessment of ecological risks. The definition of "human health" is not especially problematic for health risk assessors, and the general public places a high value on "human health" protection measures (even if there is sometimes debate about what those measures should be).[b] Assessment of risks vis-à-vis "human health" is therefore both scientifically meaningful and socially relevant. Some ecologists have defined a parallel concept of "ecosystem health,"[10,11] but the appropriateness of this concept and the means to define and measure it are controversial among ecologists,[12-15] and there is no consensus among the general public about what constitutes ecological health or in which instances, or in what forms, it must be preserved.[16]

2.1.1.1 Planning

Lacking such a clearly-defined reference point, ERA calls for an initial planning step that includes the explicit establishment of ecosystem management goals (Figure 2-1).[1] The planning process is a dialogue between risk assessors and risk managers and, where appropriate, interested and affected parties ("stakeholders"), to determine the goals and scope of the assessment. However, according to USEPA,[1] planning should be separated from the scientific conduct of the

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[a] By some definitions, inherent uncertainty is termed variability, and the term uncertainty is reserved for knowledge uncertainty.[126]

[b] While the World Health Organization has defined human health broadly as "a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity," health risk assessment as practiced by environmental agencies is concerned only with hazards causing "damage," "injury" or "harm." [2,3,127] "Human health" for risk assessors is thus the absence of these adverse conditions.

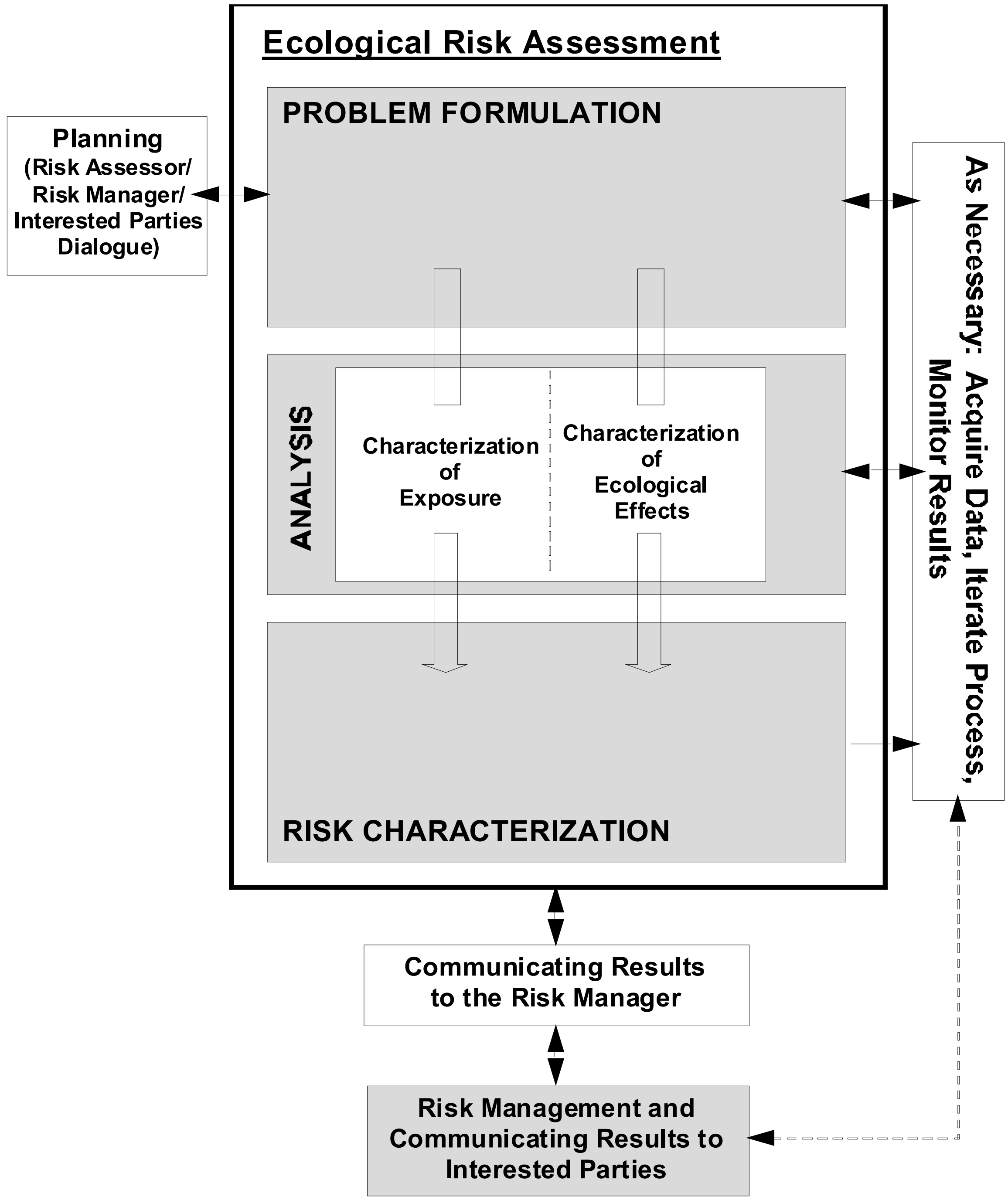

FIGURE 2-1

Framework for Ecological Risk Assessment (from USEPA[1]) risk assessment proper, to "ensure that political and social issues, though helping define the objectives for the risk assessment, do not bias the scientific evaluation of risk." This separation is consistent with a principle espoused by the National Research Council (NRC);[9] however, its appropriateness is explored further in Section 2.1.1.5 and Chapter 3.

ERA planners seek agreement on (1) the decision context, (2) management goals and objectives, and (3) information needs. Characterizing the decision context entails understanding the decisions faced by officials, groups or citizens regarding an environmental problem, as well as the public values, the legal, regulatory, and institutional factors, the geographic relationships, and the available risk management options that make up the context of those decisions. It also includes identifying risk assessors, risk managers, other specialists, and interested individuals and groups who should be involved in the planning process. Management goals are "general statements about the desired condition of ecological values of concern"[1] whereas management objectives are sufficiently specific to allow the development of measures.[17] Objectives must identify "what matters" given the decision context (in other words, what valued ecological characteristic should be protected), what protection requires, and what level of improvement, or direction of change, is to be achieved. Examination of informational needs entails determining whether an ERA is warranted and, if so, its scope, complexity and focus.[17] Suppose, for example, there were concerns over the decline of a sport fishery in a reservoir influenced by municipal effluents and agriculture. Understanding the decision context may require listing the potential regulatory or restorative actions that could be taken by officials, farmers, reservoir users and other citizens throughout the watershed; involving individuals representing each of those groups; and appreciating the values and the legal and economic interests held within each group. The management goal might be to maintain a viable sport fishery in the reservoir, and objectives might entail a listing of desirable species to be maintained.

2.1.1.2 Problem formulation

USEPA[1] defines problem formulation as “a process of generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, from human activities.” It requires (1) the identification of assessment endpoints, (2) the development of one or more conceptual models, and (3) the development of an analysis plan. Assessment endpoints operationalize the valued ecological characteristics identified in the management objectives by, first, identifying those that are both ecologically relevant and susceptible to human caused stressors and, next, selecting specific ecological entities, and measurable attributes of those entities, to embody those valued characteristics in the analysis. For example, if a management objective was to maintain a viable fishery for a list of popular recreational species, then assessment endpoints might include population size, mean individual size and recruitment for those species.

A conceptual model is “a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed.”[1] The visual representation usually takes the form of a box-and-arrow diagram illustrating hypothesized relationships between sources (human activities that produce stressors), stressors (chemical, biological or physical entities that can induce an adverse response), exposure pathways, and receptors (ecological entities that may be adversely affected). An example is presented in Chapter 5 (see Figure 5-3). Initial versions of the conceptual model for a complex problem may be overly detailed; later versions can be simplified to emphasize only those pathways that figure importantly in the analysis plan.

The analysis plan identifies those hypotheses[a] that are believed to be important contributors to risk, or that can be feasibly reduced through management efforts. The plan specifies data needs, data collection methods and methods for analysis of existing or newly collected data in order to confirm, or quantify, the underlying relationships and estimate risks.

Referring again to the reservoir fishery example, if fishery declines are hypothesized to result either from low dissolved oxygen concentrations caused by excessive nutrient inputs from municipal and agricultural sources or from agricultural pesticide use, diagrams (and accompanying text) would be produced illustrating these hypothesized sources and pathways of pollutant transport to the lake. The ecological processes specific to each pollutant, nutrient effects on dissolved oxygen levels, pesticide effects on aquatic food webs, and ultimate effects on the assessment endpoints would also be diagrammed. Following an evaluation of existing data, an analysis plan might call for the analysis of data on pesticide use in the watershed, municipal effluent characteristics, water quality in the lake and its tributaries, and fish populations.

2.1.1.3 Analysis

Analysis is "a process that examines the two primary components of risk, exposure and effects, and their relationships between each other and [with] ecosystem characteristics."[1] Exposure analysis describes sources of stressors, stressor transport and distribution, and the extent of contact or co-occurrence between stressors and receptors. Exposure analysis may be carried out using environmental measurements, computational models, or a combination of these. The product of exposure analysis is an exposure profile describing the intensity, spatial extent

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[a] Except as otherwise specified, "hypothesis" in this document refers to a "maintained hypothesis," or statement thought to be true (i.e., an assumption).

and timing of exposure. In effects analysis, the effects that are thought to be elicited by a stressor are first identified. Effects of concern are then subjected to an ecological response analysis, which examines the quantitative relationship between the stressor and the response, the plausibility that the stressor may cause the response (causality), and links between particular measures of effect and the assessment endpoints. In the sport fishery example, exposure analysis would examine the magnitude, timing and spatial dynamics of nutrient inputs; it would also characterize reductions in dissolved oxygen (DO) concentrations, since low DO constitutes a secondary stressor potentially affecting the assessment endpoints. Exposure analysis would also characterize the input, fate and transport and resulting water concentrations of pesticides used in the watershed. Effects analysis would include a literature analysis to identify the kinds of effects potentially caused by these stressors and to determine whether exposure-response relationships had been estimated for the same or phylogenetically similar species. It would also evaluate the possibility that the primary effects of one of these stressors on the food base are causing secondary effects in the assessment endpoints. Effects analysis would also examine the relative timing of exposures and observed effects of concern to determine whether there is a causal relationship.

2.1.1.4 Risk characterization

Risk characterization is the process of uniting information about exposure and effects, in order to first estimate and then describe the likelihoods of adverse effects of stressors. Risk estimates range in sophistication from simple, qualitative risk ratings (e.g., high, medium or low), used when information is limited, to comparisons of point estimates of exposure and effective level, to comparisons of probability or frequency distributions of exposure and response.

Figure 2-2 illustrates the latter case. The intensity of exposure to a stressor varies across an assessed population of individuals, and this variability is expressed as a cumulative frequency (curve on left). The fraction of individuals in a tested population that responded to a given intensity of exposure also varied (curve on right). By aligning these curves on the same exposure axis, it is shown that median exposure is below the median level of sensitivity by a relatively large margin, and that 90% of individual exposures are below a level that caused a response in 10% of individuals, albeit by a smaller margin. These data would suggest a very low level of response is expected in the assessed population, as long as the test population adequately represents the assessed population.

Risk descriptions that accompany risk estimates should discuss the adequacy and quality of data on which the assessment is based, the degree and type of uncertainty associated with the evidence, and the relationship of the evidence to the hypotheses of the risk assessment. For example, the exposure and response distributions represented in Figure 2-2 may represent inherent uncertainty, which cannot be further reduced, that is due to variability in the environmental distribution of the stressor and in the sensitivity of organisms tested. But there may be knowledge uncertainty associated with the data as well, if the number of exposure measurements or organisms tested was too low to adequately characterize the variability or if there were problems or biases associated with those measurements. There may be knowledge uncertainty concerning whether the response of the wild assessment population is similar to that of the test population, or whether the duration of the test and the endpoints examined were sufficient to characterize the possible effects. Risk descriptions should evaluate all lines of evidence, both supporting and refuting the risk estimates. They should also discuss the extent to

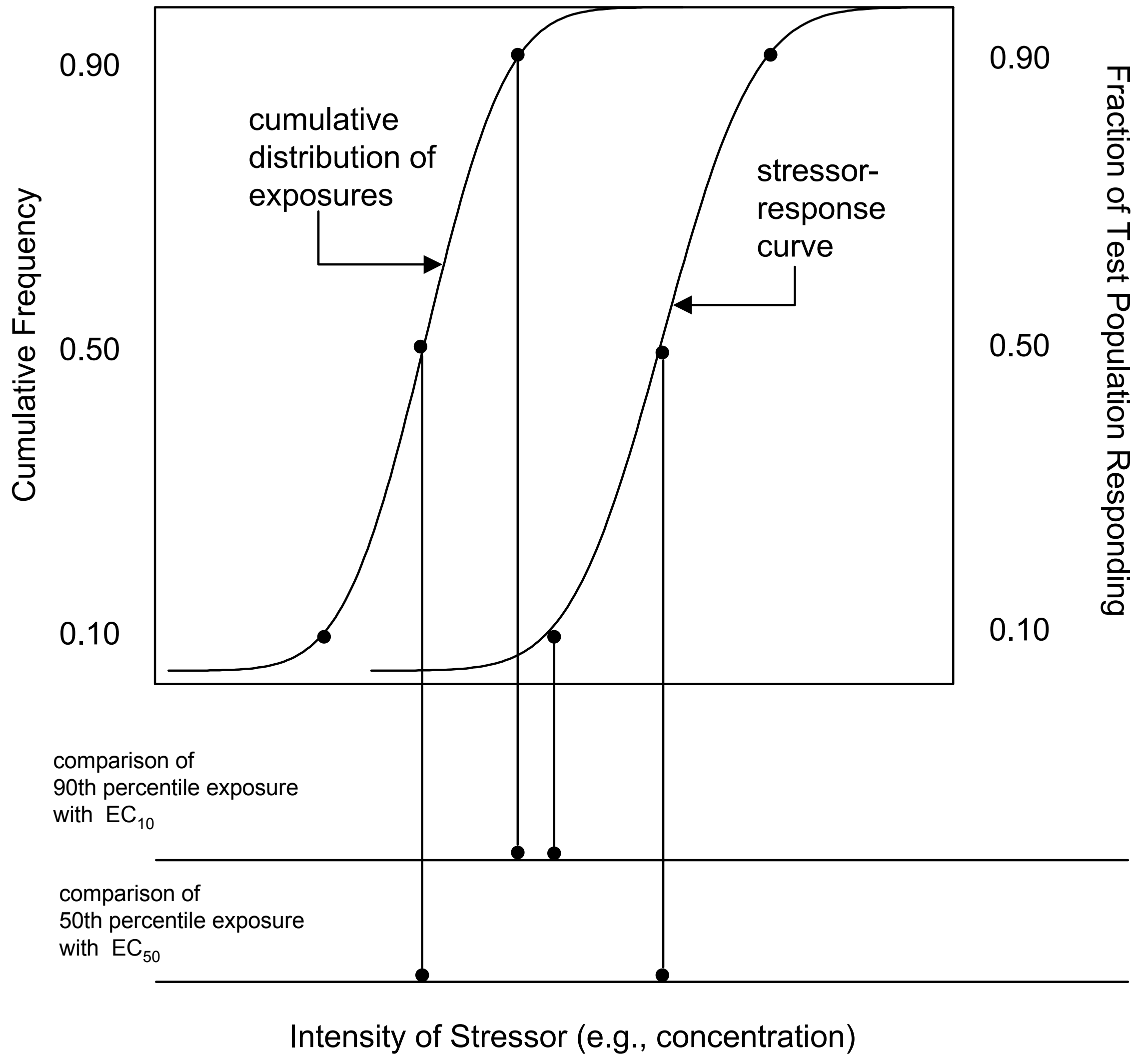

FIGURE 2-2

Estimation of risk by comparing a cumulative frequency distribution of exposure to a stressor and a stressor-response relationship; EC_X denotes stressor concentration affecting X% of test population (from USEPA[1]) which changes predicted in the risk assessment should be termed adverse, including the nature and intensity of expected effects, their spatial and temporal scale, and the potential of affected species or ecosystems to recover.

2.1.2 Critiques of ecological risk assessment

Using the steps of planning, problem formulation, analysis and risk characterization, ERA seeks to provide a concise roadmap for science-based decision support – beginning with an inclusive, policy-informed discourse, proceeding through a rigorous process of hypothesis generation, data gathering and evaluation, and leading to a set of carefully delimited statements about the probabilities of specific, adverse outcomes, to be provided to decision-makers. The process is intended to be flexible; it can employ tiers of increasing specificity (e.g., from screening-level to definitive), and sequences can be iterated as needed before proceeding to subsequent steps (see Figure 2-1).

Nonetheless, ERA has been subject to various criticisms. Some of these pertain to problems of application, others to methodology, and others to the premises underlying the role of science in decision-making. Many are centered on the treatment of scientific uncertainty, and several involve questions of whether science and policy can, or ought to, be separated. It is important to consider these issues openly when the use of ERA is contemplated for decision support – partly to be aware of the potential for misuse of the ERA process, and partly to acknowledge concerns that may be held by many stakeholders.

Some critics have charged that ecological risk assessors are prone to a rather sanguine view of the process, in which long-term laboratory tests of properly chosen sentinel species are assumed to yield results that are stable and adequately predictive of ecosystem responses (see Power and McCarty[18] and ensuing discussion).[19-21] They argue that the variability in stress-response among species and among field sites is sometimes ignored, and that biological regulatory mechanisms operating at the level of the field population or the ecosystem can confound the conventional interpretation of laboratory test results. These criticisms highlight the importance of using multiple lines of evidence (e.g., both field and laboratory observations) and making a full presentation of assumptions and uncertainties when characterizing risk, as called for in the ERA *Guidelines*.[1]

A common mistake in the analysis stage of ERA is ignoring statistical power – i.e., the probability that a given experiment or monitoring study will detect an effect if it actually exists.[22-24] If hypothesis testing fails to reject the null hypothesis (no significant effect is detected), statistical power analysis determines the level of confidence that can be placed in the negative result; when power is low, a greater need for precaution is indicated.

Where the above criticisms pertain largely to ERA methods and applications, more fundamental issues have also been raised (see especially papers from a symposium held in 1994 entitled "Ecological Risk Assessment: Use, Abuse and Alternatives"[25] and calls for the use of precautionary rather than risk-based approaches [e.g., the Wingspread Statement on the Precautionary Principle[26]]). Critics claim that (a) unintended ecological consequences of past actions demonstrate that ecosystems are too complex to be predictable under novel conditions, and (b) in view of these inherent uncertainties, it is immoral to rely upon the results of even a well-conducted risk assessment if alternative (albeit more costly) courses of action exist that appear to pose less hazard.[27,28] A related argument (see the Wingspread Statement) adds that the burden of removing uncertainty must lie with the proponent of any potentially risky action rather than with society at large. These arguments sometimes portray even the unbiased risk assessor as an enabling participant, who by virtue of his/her expertise lends a cloak of legitimacy to an intrinsically unjust process.[29] More often, the assessor is portrayed as biased (e.g., holding a narrowly reductionist worldview or having an organizational conflict of interest) or intentionally deceptive. In the end, according to this critique, ERA is at best unreliable for decision-making and at worst a tool to facilitate ecosystem exploitation.

Some of these criticisms pertain to governance structures themselves rather than to ERA per se. If indeed the validity of the governance structure underlying an environmental management effort is itself in dispute, then the trust that is necessary for an effective planning dialogue may be impossible to obtain, and ERA may be ineffective. In most cases, however, if an effective dialogue as described in the ERA *Guidelines* can be established, then many of the practical and fundamental issues that critics raise can be accommodated, even where deep-seated disagreements exist. As stated above, an effective planning dialogue clarifies the decision context, including participant values, burden of proof, institutional factors and management alternatives, and ensures that the assessment is not too narrowly conceived. Organizational interests and biases can be made clear at this stage as well. The *Guidelines* also state that the appropriateness of including stakeholders depends on the circumstances; in some cases, existing law and policy might narrowly prescribe the terms for conducting an assessment. However, it is unlikely that such a restriction ever is appropriate for assessment of problems in watersheds, where there are multiple sources and stressors, a variety of resources to protect, various regulatory authorities and incentive programs, and a need for broad community support.

In summary, through an inclusive planning dialogue and careful treatment of uncertainty, an ERA conducted according to the *Guidelines* can address many of the practical and philosophical criticisms that have been leveled against risk assessment. Further steps may need to be considered as well. Whereas the *Guidelines* argue for a strict delineation of policy and science – the planning process, where stakeholders may participate, remains "distinct from the scientific conduct of [the] risk assessment"[1] – other scientists have argued that the limits of science should be acknowledged not only at the planning stage but throughout the assessment. When risk assessors are forced to make judgments that go beyond the limits of the data, as they routinely do, they move from the realm of science into what Alvin Weinberg[30] has termed "trans-science." These judgments reflect the knowledge, experience and even cultural values of the assessor,[31] and they cannot, according to Weinberg, be viewed as free of bias. Funtowicz and Ravetz[32,33] likewise have suggested that as uncertainties, decision stakes and urgency increase, problem-solving strategies correspondingly must progress from "applied science," to "professional consultancy," to "post-normal science." Post-normal science does not pretend to be value-free or ethically neutral, and it makes use of deliberation. The NRC[34] acknowledged that deliberation, including interested and affected parties, in the problem formulation stage of risk assessment can elicit insights that would not occur to assessors and managers alone, and they called for deliberation involving decision-makers and interested and affected parties throughout the risk assessment process. The participation theme will be discussed further in Chapter 3.

2.1.3 Watershed applications of ecological risk assessment

The use of the watershed as a geographic unit for planning and management of water supply and flood control in the U.S. dates to the late 19th and early 20th centuries, but its use for ecosystem protection is more recent.[35] After the formation of the USEPA in 1970, the need for such an approach grew steadily – as environmental regulatory programs proliferated yet were spatially uncoordinated and lacked efficient mechanisms for sharing information. Also, during this period point-source pollution problems were beginning to be solved through the issuance of discharge permits, bringing to light the less tractable problems of nonpoint sources and habitat modification. Finally, in the 1990s environmental groups began to sue the USEPA over its failure to go beyond the source-by-source issuance of discharge permits, in the thousands of cases where these had proved insufficient to rectify water quality impairment. Dozens of court actions, brought under the water quality standards provision of the Clean Water Act of 1972 (CWA), required the States or the USEPA to determine, on a whole-water-body basis, the total maximum daily load (TMDL) allowable from all sources.

For these reasons, in the 1990s the USEPA began to encourage the use of a “watershed protection approach” (later termed simply the “watershed approach”) for evaluating and managing threats to freshwater and estuarine ecosystems,[36-40] and they defined a framework for that process (a discussion of this and other frameworks is presented in Chapter 3). This approach provided an effective way of spatially delimiting ecological resources and the threats to those resources, engaging stakeholders in protection efforts, and promoting management actions that were concerted rather than piecemeal. Thus, the watershed protection approach focused on goal-setting, partnerships and management. Early USEPA guidance on the approach did not describe a role for ERA; there was an emphasis on procedures for calculating TMDLs,[41] but these were aimed at determining how to meet numeric water quality standards (WQS) rather than at determining risks per se (see further discussion of WQS in Section 2.3). However, WQS do not address several aquatic ecological problems, including those due to hydrologic modification (e.g., water withdrawal, flow control, or development-related changes in runoff and recharge patterns), stream channel modification, removal of riparian vegetation, and introduction of nonnative species. Nor can they address chemicals for which no standards have been defined, indicate which of several pollutants may be causing an observed impairment, nor indicate whether a given protective or restorative measure, if implemented, will reduce the pollutant successfully. Nor can WQS adequately address problems whose severity is a function of spatial scale or the interactions of multiple stressors. Even motivated and involved teams of citizen and governmental partners can fail to achieve ecological improvements when risks in a watershed are not adequately understood. These are questions ERA is geared to address.

Therefore, ERA has a significant role to play as a tool for watershed management.[42] Five watershed ecological risk assessment (W-ERA) case studies were initiated by USEPA in 1993[43,44] and results for several of these recently have been published.[42,45-49] The case studies were initiated to evaluate the feasibility of applying the ERA process to the complex context of watershed management. Watersheds were selected for study on the basis of data availability, identification of local participants, diversity of stressors, and significant and unique ecological resources. The watersheds selected were the Big Darby Creek in central Ohio, the Clinch River Valley in southwest Virginia and northeast Tennessee, the Platte River watershed in Colorado, Wyoming and Nebraska, with special emphasis on the Big Bend Reach in south central Nebraska, the Middle Snake River in south central Idaho, and Waquoit Bay on the southern shore of Cape Cod in Massachusetts (Figure 1-1). These watersheds comprised different surface water types, stressors, scales, management problems, socioeconomic circumstances, and regions.

An initial review of progress of these assessments through the problem formulation stage[44,50] found that ERA provided formal and scientifically defensible methods that were a useful contribution to a watershed management approach. They also found that the analyses in these five cases had not been as strongly linked to watershed management efforts as would be desired. However, subsequent experiences from these assessments have suggested that following W-ERA principles increases the likelihood that environmental monitoring and assessment data are considered in decision-making.[42,51,52] The three major principles that proved most beneficial were (1) holding regular meetings between scientists and managers to establish assessment goals and to share interim findings that could be of immediate value to managers, (2) using assessment endpoints and conceptual models to understand and communicate cascading effects and identify the most significant ecological concerns, and (3) combining data from many sources into an overall analytic framework, within which multiple stressor analysis is made feasible.[42] Later chapters of this report will present the findings of economic studies that were funded in 1999 in three of those watersheds in order to further utilize the ERA results and extend their value for decision-making.

2.2 ECONOMIC ANALYSIS

This section discusses economic analysis in relationship to watershed management. As with the preceding discussion of ERA, the goal is to provide sufficient background to make the succeeding chapters understandable to the non-economist, rather than to provide a comprehensive introduction to the topic. First, it describes welfare economics as the foundation of environmental and natural resource economics, and the related concept of economic value. Next, this section introduces some tools that are used for the valuation of environmental goods and services, and some watershed-related applications of those tools. Then it introduces game theory, a set of approaches for modeling decisions that are based on economic theory. Finally, it discusses ecological economics, an emerging field that has criticized the mainstream economic paradigm, and its potential contribution to the practice of watershed analysis.

2.2.1 Welfare economics

Economists study the allocation of scarce resources across competing uses. Like time and money, the allocation of environmental goods and services entails important choices, because all wants cannot be satisfied.

Welfare economics is the study of agents who are making choices, under the given assumption that they are trying to maximize their well-being (i.e., their welfare or satisfaction, also termed utility). Economists focus on choices made by agents such as individuals or firms. They assume individuals are rational -- that is, they make choices that maximize their well-being subject to constraints on time and money -- and that firms maximize profits subject to technology or resources. These decisions are examined through marginal analysis – that is, by determining how beneficial or costly one additional unit of a good or service would be to the agent.

In an ideal market, agents' decisions will lead to an efficient outcome, or one in which all mutually beneficial trades have been made. In other words, under conditions of economic efficiency, also termed Pareto efficiency, the distribution of resources among agents is such that no one can be made better off without making someone else worse off.[a] Rarely, however, do markets achieve efficient outcomes for environmental goods and services.[53] More often, characteristics of the market or of the goods and services make trade in the marketplace difficult. Economists describe these as situations of market failure, and they may attempt to identify social arrangements, including policies and institutions, for adjusting the distribution of resources in order to improve efficiency.

Aquatic ecosystems provide many goods and services to humans (Table 2-1). Some of these, like hydropower or bottled water, are traded in markets, yet imperfections in these markets may lead to inefficiency and degradation. Others, including public goods such as recreational fishing sites and ecological services such as aesthetics or groundwater recharge, may lack markets entirely; economists refer to these as nonmarket goods and services.

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[a] It should be noted that efficient outcomes are not always fair. The concept of equity is discussed in Section 2.2.4.

TABLE 2-1

Daily's classification of ecosystem services with illustrative examples

Production of Goods	• food (terrestrial animal and plant products, forage, seafood, spice) • pharmaceuticals (medicinal products, precursors to synthetics) • durable materials (natural fiber, timber) • energy (biomass fuels, low-sediment water for hydropower) • industrial products (waxes, oils, fragrances, dyes, latex, rubber, etc., precursors to many synthetic products) • genetic resources (intermediate goods that enhance production of other goods)
Regeneration Processes	• cycling and filtration processes (waste detoxification and decomposition; soil fertility generation and renewal; air and water purification • translocation processes (dispersal of seeds necessary for revegetation; pollination of crops and natural vegetation
Stabilizing Processes	• coastal and river channel stability • compensation of one species for another under varying conditions • control of the majority of potential pest species • moderation of weather extremes (such as of temperature and wind) • partial climate stabilization • hydrological cycle regulation (mitigation of floods and droughts)
Life-Fulfilling Functions	• cultural, intellectual, and spiritual inspiration • aesthetic beauty • existence value • scientific discovery • serenity
Preservation of Options	• maintenance of the ecological components and systems needed for future supply of these goods and services and others awaiting discovery

(Adapted from Daily, GC. *Environ. Sci. and Policy,* 3, 333, 2000 and as cited in USEPA, Planning for Ecological Risk Assessment: Developing Management Objectives, External Review Draft, EPA/630/R-01/001A, June 2001.)

Further inefficiencies in the market exist because aquatic ecosystems have been used as waste receptacles; third parties are “external” to these market transactions, although they are affected by them. Consider, for example, pollutant discharges by a firm into a river that is used by downstream households for recreation; regular markets provide no mechanism to compensate these third parties for the effects of these “externalities” and are therefore inefficient.

A final type of market failure occurs when the economic agents have incomplete information, or differing information, about a good or service.[54] Information may be incomplete because not all the relationships within an aquatic ecosystem are fully known; for example, decisions to pollute or to develop may be made without full understanding of the consequences.[54] Asymmetric information may lead to strategic interaction among those involved, rather than straightforward responses based on supply and demand.[55,56]

Recognition of these kinds of market failure has led to the development of natural resource and environmental economics as specialized subfields of welfare economics. Natural resource economics examines the optimal allocation of scarce resources over time, including both nonrenewable resources (e.g., minerals) and renewable resources (e.g., fisheries and water resources).[53] Environmental economics tends to focus on two main issues: regulating pollution or damages as an externality, and valuing nonmarket goods.[57]

2.2.2 Economic value

At this point it is necessary to provide a clear definition of economic value. Freeman[58] defines economic value within the welfare economic framework. Because each individual is considered to know how well off he or she is in a given situation, and each individual’s well-being depends on both private and public goods, then economic value of any particular good should be based on the associated changes to individuals’ well-being. In some cases, markets help define economic value, but in the absence of markets, or in cases of market failure, other techniques are needed.

In either situation, economic value is defined as the maximum of something someone is willing to give up to get something else.[58] It does not need to be measured in dollars (e.g., an individual may be willing to give up the usefulness of a dam to obtain an increase in water quality and better fishing), but the dollar metric allows economists to compare trade-offs to all other goods. Willingness to pay (WTP) is a monetary measure of a welfare change or economic value; it is the maximum amount a consumer would pay in order to obtain or avoid a particular change. An alternative measure to WTP is willingness to accept (WTA), the minimum amount of money an individual is willing to take to give up some change. Both WTP and WTA measure value, but they are likely to differ for a number of reasons.[58-60] For example, they use different starting points for the initial levels of well-being (for an improvement, WTP is measured by starting at the individual's level of well-being before the improvement and WTA is calculated by starting at the individual's level of well-being after the improvement). Also, WTP is constrained by income while WTA has no upper constraint. Economists typically use WTP to value benefits because it is easier to estimate.[59]

Economic value for environmental goods and services has been separated into use and nonuse value. Use value applies when people get some satisfaction from personal utilization of environmental goods and services; use can be direct or indirect use. An example of direct use is enjoying the woods while hiking. To one who enjoys fishing for smallmouth bass, indirect use may mean valuing crayfish because smallmouth bass eat them. The idea of nonuse value, first introduced by Krutilla,[61] comes from the notion that individuals can value environmental goods and services regardless of whether they use the resource. For example, individuals in the U.S.

are willing to devote resources to protecting Brazilian habitat for the endangered, golden tamarind monkey, even though they do not ever expect to visit the area or to see the species. The total economic value for a nonmarket good or service is the aggregate of these categories of values.

Economists have developed a variety of methods for estimating nonmarket values.[58] The methods can be categorized according to how the data are generated (based on observed or hypothetical behavior).[62] Observed-behavior approaches, referred to as revealed preference methods, infer values from data on actual market choices related to the public good. Table 2-2 briefly describes four revealed preference approaches. Revealed preference approaches require market data, which limits the kinds of environmental goods that can be valued. The assumptions on which these approaches rely also affect the results. The hedonic price method, which examines the effect of differences in environmental quality on, for example, housing or job markets (Table 2-2), assumes that all buyers in the market perceive these environmental characteristics.[63]

Hypothetical approaches, called stated preference methods, use data generated by placing individuals in hypothetical choice settings. These methods are needed when no behavior can be observed (or no other market data exist to infer value), such as to estimate nonuse values or to value changes that have not yet occurred. These approaches typically use surveys that determine WTP or WTA; Table 2-2 describes two such approaches. Stated preference methods typically require more time and cost to develop and implement than revealed preference approaches, and can be subject to bias. These biases can create uncertainty about whether respondents would actually pay the amounts they indicate.

TABLE 2-2

Methods for estimating values of environmental goods and services

Method	Description	Examples
Revealed preference methods (can estimate use values only)		
Market	When environmental goods are traded in markets, their value can be estimated from transactions.	The benefits of an oil spill cleanup that would result in restoration of a commercial fishery can be projected from changes in markets for fish, before and after the spill, and their effects on fishermen and consumers.
Production function	The value of an environmental good or service can be estimated when it is needed to produce a market good.	If an improvement in air quality would lead to healthier crops, the value of the improvement includes, e.g., the reduction in fertilizer costs to produce the same amount of agricultural crops.
Hedonic price method	The value of environmental characteristics can be indirectly estimated from the market, when market goods are affected by the characteristics.	If an improvement in air quality improves a regional housing market, its value includes increases in housing value, which can be measured by statistically estimating the relationship between house prices and air quality.
Travel cost method	The value of recreational sites can be estimated by examining travel costs and time.	The value of a recreational fishing site to those who use it can be estimated by surveying visitors, to determine the relationship between the number of visits and the costs of time and travel.
Stated preference methods (can estimate both use and nonuse values)		
Contingent valuation method	Individuals are surveyed regarding their willingness to pay for a specifically described nonmarket good.	In a telephone survey, respondents are directly asked their willingness to pay, via a hypothetical tax increase, for a project that would reduce runoff, improving the health of a particular stream.
Conjoint analysis	Survey respondents evaluate alternative descriptions of goods as a function of their characteristics, so the characteristics can be valued.	In a mail survey, hypothetical alternative recreational fishing sites are described by type of fish, expected catch rate, expected crowding and round-trip distance; respondents' preferences are used to calculate value for changes in each of the characteristics.

Benefit transfer is an alternative to either stated or revealed preference methods.[60] This method estimates the value of environmental goods and services by transferring the results of previous studies at different locations.[64] For example, the value of clean water in Ohio could be approximated using a number of different studies that estimate the value of reducing nutrients in Pennsylvania waterways. Like stated preference methods, it can be used in the absence of market data, but it is less expensive to implement. However, many factors need consideration to determine whether benefit transfer will provide adequate information.[59]

To summarize, the choice of valuation technique depends on the values individuals have for the good or service (i.e., use and nonuse), the availability of appropriate data, the researcher's constraints (e.g., time and money), and the ability to minimize biases. For more detail on revealed preference methods, stated preference methods and benefit transfer approaches (such as the theory, analysis and steps), the reader is referred to Freeman,[58] Hanley and Spash[63] and Desvousges et al.[65] For additional information on estimating ecological benefits, the reader should see USEPA.[59,66]

Two of the case studies presented in later chapters of this document used stated preference techniques (Table 2-2). Chapter 4 explores the use of a contingent valuation method (CVM) model to value alternative development approaches in the Big Darby Creek watershed of central Ohio, and Chapter 5 presents a study of the use of conjoint analysis (CA) to study social trade-offs among development policies in the Clinch Valley of southwestern Virginia and northeastern Tennessee. To prepare the reader unfamiliar with those methods, Appendix 2-A discusses their differences more at length.

2.2.3 Cost-benefit analysis

Cost-benefit analysis (CBA) is the process of summing the value of the individual welfare changes, present and future, associated with a project or policy. The purpose is to assess all changes that can be feasibly measured to determine whether society gains more than it loses. If the benefits exceed the costs so that the gainers could potentially compensate the losers – this is termed the potential Pareto criterion – the project or policy is said to improve efficiency.[59,63] Under this criterion, it is considered irrelevant whether compensation actually occurs. The procedure may be used prospectively, in planning, or retrospectively, to determine if planned goals were met. CBA was originally developed to assess the net economic value of public works projects, the outputs of which usually were market goods, and the goal of which was to produce net social benefit.[58,63] Some of the earliest examples of its use were for water resource projects in the U.S.,[63,67] so the relationship between CBA and watershed management is longstanding.

Hanley and Spash[63] describe eight stages of CBA (Table 2-3). The first stage defines what is to be analyzed, to reveal how the project or policy will cause change. The next stages identify the relevant impacts and their physical characteristics, including applicable time horizons, as necessary for economic comparison. For example, if stream restoration is undertaken to improve stream ecological communities, then the time necessary to plant the riparian zone; the duration of required maintenance; the lag period for fish population response; and the type and magnitude of the response need to be determined. The process of economic valuation is next. Its purpose is to express all changes in the common metric of dollars. Where market prices of goods and service do not exist, or do not capture the full value, corrected or "shadow" prices are calculated, as further discussed below. Negative effects of the project are

TABLE 2-3 Structure of a cost-benefit analysis
1. Definition of project/policy alternatives
2. Identification of project/policy impacts
3. Which impacts are economically relevant?
4. Physical quantification of relevant impacts
5. Monetary valuation of relevant impacts
6. Discounting of costs and benefit flows
7. Applying the net present value test
8. Sensitivity analysis

Source: Hanley and Spash[63]

estimated as opportunity costs, or the lost value of a resource that cannot be used because of the project.[59,68] For example, if a firm chooses to pollute a river, an opportunity cost might be the lost value of recreational fishing. The sixth step, discounting of cost and benefit flows, is necessary when benefits and costs occur at different times, to translate all values into present value. Present values can be compared; if the net present value is greater than zero the project or policy is said to improve efficiency. If more than one project or policy is being compared, the one with the largest net present value is said to be the most efficient or provide the largest improvement in social welfare. The final stage is sensitivity analysis, which examines the uncertainty of the relevant impacts and discount rate.

2.2.4 Complementary analyses

Traditionally, a complete economic analysis is comprised of three techniques: CBA, economic impact analysis (EIA), and equity assessment.[59,69] Where CBA provides information about economic efficiency, the other two techniques examine resource distribution. These two latter types of analysis are briefly discussed in this section, as well as cost-effectiveness analysis (CEA) and natural resource damage assessment (NRDA) as they relate to CBA.

Tietenberg[68] defines impact analysis, whether environmental or economic, as a process to quantify the consequences of various actions. By this definition, it is similar to CBA and CEA; however, rather than transforming all changes into a single (dollar) metric, it simply organizes a large amount of information for decision support. USEPA[59] defines EIA as a process to examine the distribution of impacts (both positive and negative), usually by examining economic changes across a variety of economic sectors.

Fair distribution is an important goal in both welfare and ecological economics, and equity and efficiency are sometimes traded off. Because it relies on the potential Pareto criterion, CBA is not concerned with whether the potential compensation actually takes place; therefore a project by which society as a whole benefits may cause transfers of wealth, creating winners and losers. Equity assessment allows economists to understand changes in distribution of wealth due to a policy or project. According to USEPA's economic guidance,[59] the first step is to identify potentially-affected subpopulations; next steps may involve determining each subpopulation's net benefits or the distribution of the net benefits among the subpopulations. Most often, however, equity has not been a decision criterion in water resource projects, since, as long as net benefits over society as a whole exceed zero, those subpopulations experiencing positive net benefits theoretically could compensate the others. Research to investigate how winners could compensate losers may be needed to better ensure equitable outcomes.[70]

CEA resembles CBA but considers only costs. It may be used in situations where the estimation of benefits is infeasible (e.g., because of time or budget constraints) or too uncertain.[59,68] It is also used when a specific target – such as a pollution level to be achieved, or an area of new habitat to be created – has been established by policy and is not subject to an efficiency test. Given such conditions, CEA can help sort out the management alternatives. It compares alternatives by calculating a benefit-cost ratio for each, where benefits may be measured in non-monetary units though costs are measured in dollars. The benefits may also be the achievement of a pre-specified environmental target. Alternatives with high benefit-cost ratios would be preferred to those with low ratios. Note that because benefits and costs are not measured in the same units, only a ratio can be calculated, not a difference. This also suggests that only CBA provides information about economic efficiency.

NRDA is a process for economic analysis established under the CWA, the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) and the Oil Pollution Act of 1990.[71] These Acts hold liable for damages those who release hazardous substances and oil, respectively, into the environment,[72,73] and they establish trustees (i.e., officials who act on behalf of the public) responsible for recovering the damages. Damages comprise the cost to restore the injured natural resources to their baseline condition, compensation for interim losses pending recovery, and the cost of damage assessment.[71,74] The assessment methods used are similar to those employed in CBA for valuing ecological benefits; in this sense, NRDA is a retrospective application of the assessment methods, whereas project evaluation, the more routine use, normally is prospective.[58]

2.2.5 Game theory

Game theory, like other subfields of economics, is concerned with human behavior and can examine individuals interacting within a market, or in situations of market failure. Gibbons[75] defines game theory as the study of decision problems when multiple entities are involved.

Varian[56] simply calls it the study of interacting decision-makers. This discipline provides a theory of strategic behavior where an outcome depends on many individuals' strategies and the current conditions of the situation. Most commonly, games include three elements: players, strategies, and payoffs.[55,75,76]

Game situations can be modeled as cooperative or non-cooperative. In cooperative games, players can make binding agreements affecting the overall objective of those involved.[55,76] Most environmental applications deal with non-cooperative games, in which such agreements cannot occur.[55] Strategic interaction in games can be modeled as dynamic (changing over time) or static. Game theory has played an important role in analyzing externalities, bargaining, free-riding behavior (the reaping of benefits of a public good without paying), and principal-agent problems (situations where parties have incentives to hide information or actions).[55]

For example, if a government wants to regulate a firm's ability to pollute, where only the firm knows its emissions and abatement costs, the government wants to determine how the firm will react to the environmental policy. Game theory can help design a system with incentive compatibility, i.e., one in which individuals will provide truthful estimates.[55] International externalities, where one country's action affects another's welfare (e.g., greenhouse gas emissions or water diversion), are modeled as noncooperative games if no jurisdiction exists to enforce agreements.[55] On the other hand, for interstate water disputes, enforceable agreements are possible. Chapter 6 discusses the use of cooperative game models to inform an interstate water negotiation in the Platte River watershed of Colorado, Wyoming and Nebraska.

2.2.6 Ecological economics

A relatively new paradigm has developed out of the controversies of welfare economics. Many point out that the assumptions used to develop the utilitarian perspective (e.g., preferences are fixed, agents are rational) are not always accurate.[77-79] As Gowdy[80] elaborates,

> the focus of most economists on markets and market solutions, to the exclusion of the behavior of actual ecosystems and actual human behavior, has been at least partially responsible for a variety of wildlife policy failures, from forestry to fisheries to the protection of endangered species.

Ecological economists contend that a transdisciplinary approach, spanning economics and natural science, is needed to address environmental problems.[81] Costanza describes opposing assumptions about economic growth: technological optimism (which he represents as the economists' perspective) assumes unlimited economic growth because of human ingenuity; technological pessimism (the ecologists' view) holds that technology cannot abrogate the constraints of resources and energy and that economic stagnation is inevitable.[82] He concludes that technologically pessimistic policies should be pursued, and that ecological economic research should compare pessimistic to optimistic policies and work to reduce uncertainty regarding the effects of technology. Sahu and Nayak suggest that ecological economics is not constrained by the mechanistic assumptions of welfare economics, but uses a systems approach.[83] Whereas environmental and natural resource economists define the environment as a part of the economy (the environment is an asset), ecological economists define the economy as a part of the environment.[84] Therefore, ecological economists may use the tools of conventional economics but they also believe that new approaches are needed to answer some environmental questions.[81] In valuation, for example, ecological economists place more emphasis on the physical characteristics and ecological health of the system, which may not be captured by values elicited from individuals.[85]

Daly[84] defines the goals of ecological economics as efficient allocation, equitable distribution, and sustainable scale. Two of these goals have been mentioned previously as important to welfare economics. The third relates the "physical volume of throughput," to "the carrying capacity of the environment over time;"[84] it acknowledges that excessive economic growth can cause environmental destruction.[85]

According to Tacconi,[86] the driving force of ecological economics is analysis related to describing and achieving sustainability. Toman et al.[87] define the central issue of sustainability as the well-being of future generations, subject to constraints imposed by the functioning of the natural environment.

2.2.7 Applications of ecological economics

Ecological economics does not offer a single, theoretically integrated and widely accepted analytic paradigm similar to CBA. Much of its contribution has been in the form of a wide-ranging critique of welfare economics, some of it aimed at establishing an alternative moral-philosophical framework having sustainability (rather than social welfare as earlier defined) as the objective. At the same time, the techniques of environmental and resource economics are not necessarily rejected. At some risk of over-generalization, the ecological-economic critique may be summarized as calling for the increased use of participative processes; a greater focus on equity; the integration of multiple scientific paradigms and methods; the evaluation of multiple objectives; and explicit recognition of biophysical processes and limits.

Several analytic techniques that employ a biophysical constraint have been, or could be, applied in a watershed setting. A broad family of methods has treated energy as limiting for all meaningful work and therefore useful as a biophysical and economic least-common-denominator. This premise has been applied with various forms of energy: (a) the energy used to produce goods and services in national economies ("embodied energy");[88] (b) the solar energy that is captured by living and nonliving earth systems, is transformed (intensified) by physical or biological processes, and represents the "real wealth" of both ecosystems and human economies ("emergy");[89] and the energy that is available ("exergy") or unavailable ("entropy") to do work in a process or system.[90,91] Each of these approaches assumes the goal is to maximize useful energy rather than welfare or utility. Their proponents suggest that policies diminishing available energy should be avoided, even if they appear to increase social welfare. Their appeal is that energy, unlike utility, can be comprehensively estimated, even for nonmarket goods, and is subject to accounting under the laws of thermodynamics. However, these approaches have long been criticized by welfare economists as not being able to address the scarcity of some resources (e.g., primary minerals)[92] and as simply ignoring the supply and demand principles by which economic systems really operate.[93]

"Ecological footprint" methods consider the ecological services provided by earth's terrestrial or marine ecosystems as necessary to support life and economic activity but limited by the total area of those ecosystems.[94,95] These techniques examine the areas required to support existing patterns of resource use (including, e.g., the appropriation of productive area of poorer countries to support consumption patterns of richer countries) to determine whether uses are equitable and sustainable, and to compare alternative resource use scenarios. Critics have described the method as powerfully illustrative, since the ecological demands of economic activity often go unrecognized, but also as overly simplified, technologically pessimistic and biased against trade.[96] These criticisms suggest that biophysically-based methods, while providing useful insights, should not be used uncritically as a substitute for welfare-based methods.

Other approaches have not sought to establish an alternative economic basis but rather to link abiotic, biotic and economic models in order to simulate feedbacks within ecological-economic systems. Biophysical constraints are achieved only by ensuring that the future welfare effects of anticipated ecological changes can be taken into account when policies are designed. For example, Costanza and coworkers[97-100] developed the Patuxent Landscape Model to determine the effects of societal activities, especially land use and agriculture, on aquatic biological endpoints and land values in the Patuxent River watershed. This effort has attempted to integrate models of land use conversion, agricultural practice, stream hydrologic and ecological processes, and ecological succession/habitat type, in a spatially explicit simulation framework. Approaches of this type inform welfare-based analysis with best-available ecological modeling methods.

2.3. ECOLOGICAL AND ECONOMIC ANALYSIS FOR WATER QUALITY STANDARDS

Mechanisms for safeguarding aquatic ecological resources under the CWA are aimed at providing "protection" wherever it is "attainable".[a,101] ERA procedures have been used to determine what measures are protective and whether they are physically attainable, whereas economic analyses have been used primarily to determine what is cost-effective and financially attainable. CWA language seemingly has left little room for weighing the benefits of protection against its costs, and therefore the integration of ecological and economic analyses in CWA programs has been limited.[b] However, there have been recent calls for increased flexibility in the use of benefits analysis.

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[a] Section 101(a) (2) of the CWA
[b] Exceptions include regulatory development in support of effluent guideline limitations (e.g., the development documents for Final Concentrated Animal Feed Operations[128]).

WQS underpin several important regulatory protections for U.S. waters. In addition to the WQS program itself, the CWA authorizes regulatory programs for the establishment of national effluent guidelines for specific industries, and facility-specific effluent permits and TMDLs. Effluent guidelines are designed based on available technology and its cost-effectiveness, but permit programs rely on water quality-based limits, in addition to technology-based limits. Effluent guidelines set the minimum performance in permits for a large number of industrial facilities. For facilities not covered by effluent guidelines, permits are based on technology performance, usually evaluated by the permit writer's best professional judgment. However, if WQS are not met, these permits can then be tightened or TMDLs can be required. Therefore the ecological and economic bases of WQS are worth examining.

2.3.1 Water quality standards and ecological risk assessment

Under the CWA, States, Tribes, and Territories with approved WQS programs must establish designated uses for their water bodies. Uses are designated – such as use by aquatic life, use for fishing and fish consumption, use as a drinking water supply, use for full or limited body contact recreation – for which water and habitat quality are expected to be suitable. Designation for use by aquatic life, which requires ensuring conditions suitable for "the protection and propagation of fish, shellfish, and wildlife,"[101] normally is required for all waters, and subcategories may be established where different aquatic communities have differing water quality or habitat requirements. For example, coldwater communities typical of some higher altitude or headwater streams have more stringent requirements for DO, temperature and suspended solids than nearby warmwater communities, and thus subcategories for warmwater and coldwater aquatic life use are often established. Designated uses must protect 'existing' uses (uses that have existed at any time since 1975) and should be attainable.[a]

Ambient water quality criteria (AWQC) for specific pollutants, which are scientifically-derived criteria, are then adopted to protect the designated uses. The designated uses and corresponding criteria, taken together with provisions that prevent the degradation of high quality waters, constitute the WQS for a given water body. WQS are proposed by states and approved by USEPA. They are used as a basis for setting allowable pollutant levels for point-source discharges such as from publicly-owned treatment works (POTWs), industries, combined sewer overflow (CSO) outfalls, and concentrated animal feeding operations (CAFOs) exceeding a certain size. WQS and criteria are considered to be, respectively, the regulatory and scientific foundations of programs established under the CWA to protect U.S. waters.

Even if individual dischargers are substantially in compliance with discharge limits, WQS may be violated if there are unregulated point sources or nonpoint sources such as urban or agricultural runoff (which typically are not regulated) or if the accumulation of permitted unregulated and background pollutant loads exceeds a water body's assimilative capacity. If regular violations for one or more pollutants cause the water body to be listed as impaired, the state or USEPA must conduct a study of the drainage area of the affected water body, determine the TMDL from all sources that can be assimilated without a violation of standards, and then revise all discharge permits accordingly (or possibly require control of nonpoint sources). WQS also play an important role in nonregulatory CWA programs; impairment as evidenced by WQS violations is taken into account in the targeting of Federal funds used by States or Tribes for water quality improvement projects.[102,103]

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[a] This is according to criteria and definitions in 40 CFR 131.10 (g) and 131.10 (d)

AWQC, the scientific component of WQS, are established or recommended by USEPA but may be modified by the States to reflect site-specific conditions; separate AWQC are designed for protection of aquatic ecosystems and human health. Those intended for ecosystem protection include aquatic life criteria for many toxic contaminants, clarity, DO, and nutrients.[104] They also include biological criteria (or "biocriteria"), which evaluate the condition of aquatic ecological communities. The principles of ERA play an important role in the determination of each; some are based on a characterization of stressor-response relationships and others on statistical comparison to a reference condition.

Using the terminology of ERA, AWQC derivation procedures for toxic contaminants and DO have an implicit management goal of protecting aquatic communities from adverse effects of specific chemical stressors and a management objective of limiting those stressors so as to prevent the occurrence of acute or chronic effects in 95% of aquatic taxa.[105,106] The assessment endpoint therefore is the viability of 95% of species; measurement endpoints include survival, growth, biomass and fecundity of individuals of the species tested, typically in laboratory exposures. Stressor-response assessment procedures for AWQC involve constructing species-sensitivity distributions (SSDs) of the results of acute or chronic tests for a variety of fish and invertebrate taxa, in order to estimate the concentration corresponding to the fifth percentile of tested species' responses.

Biological criteria evaluate aquatic communities themselves rather than aquatic stressors. Typically they consist of multimetric indices such as the Index of Biotic Integrity (IBI)[107] or Invertebrate Community Index (ICI)[108] which are adjusted to fit regional conditions. Biocriteria are calculated from biological survey data in order to evaluate the integrity of a water body's fish or macroinvertebrate assemblages, through comparison to a "reference" or minimally impacted condition.[109] The indices are aggregates of individual metrics that quantify the presence, abundance or condition of particular species or groups of species that have been found to be either sensitive to or tolerant of various classes of stressors. Reference conditions are usually determined by identifying reference sites (sites judged to be minimally impacted) for a given region. Biocriteria calculation methods are regionally adjusted so as to maximize their ability to discriminate reference sites from sites in the same region that are known to be impacted by human influence; sites scoring lower than a reference score are assumed likely to be suffering one or more of those impacts.

The management goal implicit in the use of biocriteria in WQS is the protection of aquatic communities from any human-induced stressors, and the management objective is the prevention of human-induced impacts on fish or invertebrate community integrity. The assessment endpoint therefore is community integrity, which may be defined as degree of similarity to "the most robust aquatic community to be expected in a natural condition."[109] Measurement endpoints are the component metrics, determined using standard biosurvey methods.[109] As a result of the use of indices such as IBI and ICI in WQS programs, their use in water quality monitoring programs is becoming more widespread and the data frequently are available for use in W-ERA. Because IBI and ICI are important in the case studies presented in Chapters 4 and 5, additional information on their derivation is presented in Appendix 2-B.

Nutrient criteria may be based on stressor-response relationships, similarity to reference conditions, or a combination of the two. USEPA derived a set of recommended nutrient criteria for regions of the U.S.[110] by statistically defining a reference condition, but the Agency also suggested using stressor-response relationships as an alternative basis.[111] For example, Ohio EPA employed an IBI score as the response endpoint in stressor-response analyses for total nitrogen (TN) and total phosphorus (TP) that formed the basis for recommended nutrient criteria[112] and for the TP target of a draft TMDL.[113]

When WQS are incorporated into discharge permits, permit limits are designed using exposure assessment methods to ensure that ambient exposures, beyond an immediate mixing zone, will be low enough to avoid acute or chronic toxicity, except under extreme low-flow conditions.[114] TMDL targets frequently are based on AWQC that have been derived from stressor-response analyses. Waste loading allocations for particular sources are derived using principles of exposure assessment so as to achieve the target (i.e., attain acceptable risk) under design conditions (such as high-flow or low-flow, depending on the nature of the source).

2.3.2 Water quality standards and economic analysis

In general, WQS are based on a level of water quality that provides for the protection of aquatic life (i.e., propagation of fish, shellfish, and wildlife) wherever it can be reasonably attained,[101] not wherever it can be shown to provide net economic benefit. The CWA provides limited basis for economic analysis in conjunction with WQS programs.[101,115] If all technology-based effluent limits for point sources as well as cost-effective and reasonable best management practices for nonpoint sources have been fully implemented and a water body still has not attained the designated use, a state or tribe may conduct a use attainability analysis (UAA) to determine if the use should be removed or a variance (i.e., a temporary suspension of a water quality standard without removing the use) should be granted.[101] The UAA must show that the designated use is not attainable based on any of several grounds, one of which is that the installation of further controls (i.e., going beyond the technology standard) would result in “substantial and widespread economic and social impact.”[a] USEPA guidance recommends that

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[a] See 40 CFR 131.10 (g)

determination of “substantial” impact be based on the financial burden to affected households (for a facility that is publicly owned) or to private-sector entities of installing additional pollution controls; “widespread” impacts are those involving relatively large changes in socioeconomic conditions throughout a community or surrounding area.[115] Conversely, where water quality is higher than required to meet designated uses, CWA antidegradation provisions prevent the issuing of any permit that would result in a significant lowering of water quality unless necessary to allow an “important” economic or social development; an “important” development is one that would have “significant” and “widespread” impacts if foregone.[115]

USEPA also performs economic analyses of WQS. Cost analyses of federally implemented regulations are required under Executive Order 12866[116] and the Unfunded Mandates Reform Act,[117] and depending on the magnitude of the Federal action, a CBA may also be presented. For example, the USEPA performed an economic analysis of the California Toxics Rule which established numeric water quality criteria for toxic pollutants necessary to meet the requirements of the CWA.[118] Even in so doing, however, USEPA does not make the promulgation of its WQS-related rules subject to an economic efficiency test (i.e., a determination of whether benefits exceed costs), nor have states, tribes, or territories relied on such a test for WQS. A 1983 proposed revision of WQS regulations that would have allowed CBA to serve as a basis for changes in designated uses was discarded following public comment. In spite of previous regulatory language that required states to “‘...take into consideration environmental, technological, social, economic, and institutional factors’ in determining the attainability of standards for any particular water segment,” the agency recognized “inherent difficulties” in balancing costs or benefits with achievement of CWA goals.[119] USEPA *Interim Economic Guidance for WQS* allows CBA to be presented as part of an economic impact analysis for UAA but suggests that the determination for assessing benefits be coordinated with USEPA regional offices.[115]

Comprehensive efforts to integrate ecological and economic analyses have been rare due, in part, to existing policy. In most cases, ecological analyses determine what measures are protective and physically attainable, and separate economic analyses determine only what is financially attainable. For example, where designated uses are not being attained, stakeholders may be engaged in seeking least-cost mechanisms for meeting a TMDL target, but stakeholder preferences with respect to the ecological or other benefits of attainment normally do not play a role in identifying the target, or in downgrading the use. However, the NRC[23] has criticized this approach to WQS as "narrowly conceived" and has suggested that a "broadened socioeconomic benefit-cost framework" be employed for use designation. Novotny et al.[120] recommended the use of CBA in UAA in cases where "the nonmarket impacts (especially on water quality benefits) are likely to be large or the costs of incremental benefit very large," in spite of a lack of guarantees that USEPA reviewers would accept such an analysis as persuasive.

Furthermore, stakeholder preferences come into greater play wherever the protection of water quality is dependent on the integrity of riparian systems and adjacent uplands – especially in headwater systems. The CWA affords little Federal authority for controlling the physical modification (other than dredging or filling) of streams or riparian systems or for the control of nonpoint source (NPS) pollution resulting from upland land uses. Headwater systems, including intermittent or ephemeral streams, while of critical ecological importance,[121] are also very numerous and highly subject to disturbance and may need to be protected through approaches involving public cooperation and evaluation of benefit. For example, the Kansas Legislature[122] has mandated that certain types of low-flow or intermittent streams be entirely exempted from

CWA requirements except on those stream segments where the economic efficiency of regulation can first be demonstrated. In Ohio, although the applicability of the CWA to headwater streams has not been questioned, a need for stakeholder input as to the appropriate level of protection is acknowledged.[123]

2.4 THE NEED FOR INTEGRATION

Risk and economics are unavoidably linked. In the post-*Silent Spring* era, U.S. society entered into a number of social contracts that arguably combined elements of bold foresight and naïveté – foresight with regard to the importance of reducing ecological risks, but naïveté with regard to scientific nuance and cost. The 1973 Endangered Species Act required Federal agencies to "insure that any action ... is not likely to jeopardize the continued existence of any endangered species or threatened species..." before the sheer numbers of endangered and threatened species and their potentially overwhelming habitat protection or restoration costs were well understood. Consider, for example, the substantial costs and far-reaching social disruption that would be required to restore some endangered salmon runs in the Pacific northwest.[124] Similarly, the 1972 CWA established as a goal "restoring and maintaining the chemical, physical, and biological integrity of [the] Nation's waters" and called for achieving a level of water quality that provides for the protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water, "wherever attainable" (33 USC 1251) well before TMDL lawsuits would require that longstanding water quality impairments be addressed and the lion's share of blame would shift from big industry and sewage treatment plants to agriculture and urban sprawl. There is now a wider recognition that reducing ecological risks is quite costly, and that its costs are paid not only by big, discrete polluters but by society at large.

Moreover, risks, as humans define them, have an economic dimension. This does not imply that ERA should be limited by economics, or serve only as input to economic analyses, but rather that any risk humans can recognize has economic implications. By definition, a risk entails a probability of an “adverse” effect, or an effect that is contrary to what is desired. Therefore, risk is defined with respect to human preference. Even in those cases where norms or standards have been established by statute or regulation, subjective interpretation is often needed. As stated earlier, it is not possible to precisely define terms such as “integrity” with reference to ecosystems, and the “attainability” of a level of water quality is usually a function of cost. In many instances, USEPA regulatory programs have been required to codify a particular interpretation of these normative terms. Whenever it is allowable and practicable, however, determining the preferences of interested and affected individuals can be a means to identify the best alternatives and to ensure broadly based support for management efforts.

Since people’s information about risks is usually incomplete, technical information about risk plays an important role in informing those preferences. Furthermore, the form of the technical information is critical. Compendia of monitoring data, problem reports or expert opinions can all prove misleading because they do not provide for the rigorous and systematic determination of, e.g., objectives, causative agency, and the probabilities and uncertainties associated with projected outcomes. Risk characterization, the last step in ERA, links each of these elements in careful, informative statements. ERA is needed if economic analysis of complex ecological problems is to be done.

Just as risks have an economic or preferential dimension, so decisions about actions to reduce risks always entail trade-offs. This interrelationship of information, preferences and effective management argues for the thorough integration of ERA and economic analysis. It should be obvious, furthermore, that an approach in which the disciplines are compartmentalized rather than integrated will invariably lead to an analysis of poorer quality. Such an approach would assume that the natural and social sciences do not bring differing lenses to the understanding of goals and problems, and that the analytical requirements of each are mutually grasped without difficulty. In fact, the fundamental relationship between the social and natural sciences has long been a matter of philosophical dispute,[125] and while dialogue between economists and ecologists has dramatically increased in recent years it still must be assumed that, in any new circumstance, conscious effort will be required to establish mutual understanding between the disciplines and a concerted approach to environmental problem-solving.

When is integrated analysis needed? ERA often is needed to determine the likely ecological responses to proposed management actions, and economic analysis often is needed to interpret those ecological changes, and other changes, in terms of human well-being – so that decisions are effective and beneficial. Whenever both ERA and economic analysis are needed to address a watershed management problem, the analytic processes should be undertaken in an integrated fashion. Here, the term 'integrated' does not necessarily imply that any distinction between the respective sciences is erased, or that either loses its essential character. It does imply that these analytic processes will be mutually informed and fully coordinated. The alternative, a piecemeal or haphazard process, is unlikely to serve decision-makers or stakeholders as well.

To accomplish integration in practice, extensive interaction is needed between the disciplines as well as with others who have relevant knowledge or a stake in solving the problem. The form these interactions should take will vary according to the circumstances, but experiences from the field of environmental management can be drawn upon to identify certain principles, and sequences of events, that help determine success. Chapter 3 will examine those experiences and present a conceptual approach for integrating ERA and economic analysis in the management of watersheds.

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APPENDIX 2-A

DISCUSSION OF STATED PREFERENCE METHODS USED IN TWO CASE STUDIES

This appendix discusses the differences between two stated preference methods used for valuing environmental goods, the contingent valuation method (CVM) and conjoint analysis (CA). CVM was used to value alternative development scenarios in the Big Darby Creek watershed of central Ohio (Chapter 4), and CA measured the trade-offs among development policies in the Clinch Valley of southwestern Virginia and northeastern Tennessee (Chapter 5).

CVM measures value directly by asking respondents' their willingness to pay, using a specified payment vehicle (e.g., a change in the electric bill or in taxes), to avoid or obtain a particular change. The question format could be open-ended (i.e., how much are you willing to pay ...?) or dichotomous-choice (i.e., would you be willing to pay $X amount: yes or no?). Mitchell and Carson[1] describe CVM as a "versatile tool for directly measuring a range of benefits for a range of goods consistent with economic theory." Unlike revealed preference techniques, which are limited to valuing existing goods at existing quantity and quality levels, CVM can be used to measure both use and nonuse values of goods that may not presently exist. As a result of compensation claims associated with the Exxon Valdez oil spill in Prince William Sound, Alaska, the National Oceanic and Atmospheric Administration (NOAA) convened a panel to conduct hearings on the validity of CVM.[2] The panel established rigorous guidelines for legally admissible studies. Nonetheless, the method remains controversial among some economists because of its hypothetical nature. Several potential biases have been identified,[3,4] and CVM models have had a mixed performance when subjected to internal and external validity tests.[5,6]

Whereas CVM typically measures the value of a good as a whole, CA induces respondents to evaluate alternatives as a function of their attributes, so that the attributes can be individually valued.[7-9] For example, a respondent may be asked to state a preference (and perhaps to rate the strength of the preference) between alternative streams for fishing. The streams are said to vary as to the type of fishing, expected catch rate, expected crowding, expected weather, and round-trip distance.[10] One attribute, in this case driving distance, usually is either a cost or a proxy for cost to allow estimation of WTP. By choosing one alternative, the respondent reveals a (strength of) preference for that particular bundle of attribute values vis-à-vis the others presented. By presenting a series of choice sets in which these attribute values are varied, respondent preferences can be disaggregated and the contribution of each attribute to the combined preference determined.

Environmental management alternatives and their fiscal, social and ecological results also occur as bundles in the real world and can be analyzed using CA. The technique has been used in environmental applications where attributes are cardinal (e.g., travel distance) or class (e.g., terrain type) variables associated with the economic and environmental elements of a choice, such as a choice of recreational opportunity[11-13] or electricity generation scenario.[14] If the key features, both ecological and nonecological, that define each alternative can be expressed by the selected attributes, then CA can be used to quantify the key sources of stakeholder preference and to inform the design of an optimal alternative.

The multiattribute choice process employed in CA could avoid or reduce certain biases associated with the bid process in CVM, especially if the choices presented were meaningful and plausible to survey respondents.[15] Potential difficulties with such an application include: (1) the difficulty of constructing choice sets that encompass the needed range of potential management options and outcomes; (2) the potential for confusing or fatiguing respondents if too many attributes or choice sets are presented; and (3) a lack of experience in applying the method to evaluate indirect or nonuse values.

CA is similar to CVM, and therefore some of the same benefits apply, including the ability to value goods that have not been observed yet (e.g., impacts of global climate change), as long as they can be described adequately to the respondent. But whereas CVM results apply only to the scenarios or goods described, CA results can be extrapolated to any good within the range of attribute values used; even a good that was not specifically tested. It also avoids some of the problems of dichotomous-choice CVM such as yea-saying (i.e., bias toward agreement). However, CA has not been subjected to the same scrutiny as CVM, questionnaire design is difficult, and the optimal design is unsettled.[15]

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APPENDIX 2-B

USING MULTIMETRIC INDICES TO DEFINE THE INTEGRITY OF STREAM BIOLOGICAL ASSEMBLAGES AND INSTREAM HABITAT

To determine if a stream provides a suitable environment for a robust biological community, measurement of a set of chemical and physical water quality parameters (e.g., toxics, dissolved oxygen, temperature) is not sufficient. A chemical that is present, but not on the monitoring list, may be affecting the stream community, and episodic exposures, which are difficult to detect without continuous sampling, can also cause long term effects. Even high quality water can fail to support robust communities if other factors affect the stream environment. The physical habitat of the stream may have been altered (such as by channelization) in a way that removes instream cover or substrate needed by organisms, and barriers such as low-head dams may prevent migration or recolonization. Changes in stream hydrology that result from watershed development or flow diversion can create flow conditions that degrade the instream environment as well.

A goal of the Clean Water Act is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters."[a] The term "biological integrity" implies a concept of wholeness that encompasses more than water quality alone. Indeed, to determine whether a stream biological community is flourishing as expected, it makes sense to measure the community itself. However, because biological communities are both complex and variable over space and time, the list of aspects that could be measured is long, and the measurements are not meaningful without interpretation. To establish an operational definition, various aggregate indices have been designed that measure selected ecological parameters and express some aspect

[a] 33 USC 1251 (a)

of “integrity” (e.g., of the fish or invertebrates assemblages present, or of the instream habitat) on a simple numerical scale. While the concept itself remains controversial,[1] and some argue that in the aggregation of measures, information useful for assessment is lost rather than gained,[2] the approach has gained sufficient acceptance to become widely used in environmental monitoring and regulation.[3] Since watershed ecological risk assessments often must rely on the data that are available, whether or not they are ideal, their application in stream assessments is common. Four such indices that are referred to in later chapters of this document are briefly described here. Methods used for computing an index vary regionally; they are modified to fit regional ecological conditions. This description relies on methods used by the Ohio Environmental Protection Agency (OEPA);[4] methods applied in other locations, while not identical, are similar.

It should be noted that indices of biotic integrity are not necessarily useful for the study or management of rare species. Although Karr and Chu state that the explicit inclusion of threatened or endangered species in an index can improve their management,[1] bioassessments that are conducted for routine monitoring of stream condition may not have the spatial or temporal intensity needed to detect them. Therefore, the indices, like those used by OEPA, may not be designed to respond to the presence or absence of rare species. Furthermore, a low score on one metric that is due to the absence of a rare species could be masked by high scores on other metrics.

Another potential weakness of integrity indices is that the choice of sampling techniques may be taxonomically limiting. For example, the Invertebrate Community Index, described below, relies heavily on artificial substrates and its metrics mainly reflect organisms that colonize those substrates. As a result, the presence and diversity of noninsect taxa such as crustaceans and mollusks, many of which are sensitive to human disturbance (e.g., see Chapters 4 and 5), are poorly reflected in the index.

Index of Biotic Integrity (IBI)

The IBI, originally developed by Karr,[5] expresses the status of the stream fish assemblage in a given location at the time of sampling. A stream reach of a given length is sampled by electrofishing techniques, and captured fish are identified to the species level.[6] To compute a set of 12 metrics, species are categorized into various groupings including taxonomic family, tolerance to pollution, feeding type, breeding type, and whether indigenous or exotic (Table 2-B-1). Visible skin or subcutaneous disorders are also recorded; these include deformities, eroded fins, lesions/ulcers and tumors. For each metric, a score of 5, 3 or 1 is assigned according to whether the sample approximates (5), deviates somewhat from (3) or strongly deviates from (1) the reference value, or that value expected under minimally impacted conditions. For most metrics, the reference value is scaled according to drainage area (i.e., the area of the watershed above the point sampled), since fish assemblages in larger streams tend naturally to be more diverse. The index is a sum of scores of the individual metrics, with a maximum score of 60. The interquartile range (25^{th} percentile - 75^{th} percentile) of IBI for wadable, warmwater reference sites in Ohio is 38-50.[a,4]

Modified Index of Well-Being (MIwb)

The Index of Well-Being, developed by Gammon[7] and modified by OEPA,[4] also expresses the status of fish assemblages. It uses the same sampling data as required for the IBI but also requires determination of the total weight of each species in the sample. The index is computed as follows:

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[a] Wadable streams are those that can be sampled by personnel walking in the streams, but do not include headwaters streams (drainage area < 20 mi^2). Warmwater streams, which include most streams in Ohio, are those not capable of supporting coldwater fauna such as trout.

TABLE 2-B-1 Individual metrics constituting two indices of biological integrity used by the Ohio Environmental Protection Agency		
Metric #	Index of Biotic Integrity (IBI)[a]	Invertebrate Community Index (ICI)
1	Total number of indigenous fish species	Total number of taxa
2	Number of darter species (Percidae)	Number of mayfly taxa (Ephemeroptera)
3	Number of sunfish species (Centrarchidae)	Number of caddisfly taxa (Trichoptera)
4	Number of sucker species (Catostomidae)	Number of true fly taxa (Diptera)
5	Number of pollution intolerant species	Percent mayflies (Ephemeroptera)
6	Percent abundance of tolerant species	Percent caddisflies (Trichoptera)
7	Percent abundance of omnivores	Percent Tanytarsini midges
8	Percent abundance of insectivores	Percent other true flies and non-insects
9	Percent abundance of top carnivores	Percent pollution tolerant organisms
10	Total number of individuals	Number of EPT taxa[b]
11	Percent lithophils (species requiring clean gravel/cobble for spawning)	
12	Percent with deformities, eroded fins, lesions and tumors	

[a] Metrics listed are for wadable, nonheadwaters sites. For other sites, some metrics differ.
[b] EPT = Ephemeroptera (mayflies), Plecoptera (caddisflies) and Tricoptera (stoneflies). Index is determined only from sampling of natural, not artificial, substrates.

$$\text{MIwb} = 0.5 \ln N + 0.5 \ln B + \overline{H}\text{(no.)} + \overline{H}\text{ (wt.)}$$

where:

N	=	relative numbers of all species excluding species designated “highly tolerant”
B	=	relative weights of all species excluding species designated “highly tolerant”
$\overline{H}$ (no.)	=	Shannon diversity index based on numbers
$\overline{H}$ (wt.)	=	Shannon diversity index based on weight

and the Shannon diversity index is computed as:

$$\overline{H} = -\Sigma \frac{(n_i)}{N} \ln \frac{(n_i)}{N}$$

where:

n_i	=	number or weight of the ith species
N	=	total number or weight of the sample

The interquartile range of MIwb for wadable, warmwater reference sites in Ohio is 8.3-9.4.[4]

Invertebrate Community Index (ICI)

The ICI was developed by DeShon and others to determine the condition of the benthic, or bottom-dwelling, invertebrate assemblage.[4,6,8] Where there is sufficient stream flow, a device consisting of a series of hardboard plates, spaced along an eyebolt, is submerged in the stream and allowed to be colonized for a period of six weeks during the summer months. It is then collected for laboratory enumeration and identification of the attached organisms. To augment observations from the artificial substrates, a net is used to sample organisms occurring on natural substrates. Where the artificial substrates cannot be used, the natural substrates are sampled more extensively. When possible, individuals collected are identified to species, but sometimes identification is only to the genus or a higher level. As with IBI, species are categorized into groups for calculation of the index. The ICI is composed of 10 metrics (Table 2-B-1) that are scored as either 6, 4, 2 or 0 according to a relationship that varies with drainage area. These relationships are more complex than those for fish. For example, diversity of certain groups first increases and then decreases as drainage area increases. Like the IBI, the highest possible score is 60. The interquartile range of ICI for reference sites in Ohio where artificial substrates could be used is 36-48.[4]

Qualitative Habitat Evaluation Index (QHEI)

The QHEI evaluates physical characteristics of stream habitats that are important to fish and invertebrate communities.[6,9,10] Six principal metrics compose the index, each having two to five constituent measures (Table 2-B-2). The metrics describe the material covering the stream bottom (substrate), areas where fauna can hide (cover), complexity and stability of the stream channel (channel quality), naturalness and stability of the streamside environment (riparian/erosion), variety of instream habitat types such as riffles, runs and pools (pool/riffle), and steepness of the stream in the direction of flow (gradient). The maximum score is 100. The interquartile range of QHEI for wadable, warmwater reference sites in Ohio is 68-78.

TABLE 2-B-2

Primary and secondary metrics constituting the Qualitative Habitat Evaluation Index (QHEI) used by the Ohio Environmental Protection Agency

Metric	Score
Substrate	*≤ 20*
Type	0 – 20
Quality	-5 – 3
Instream Cover	*≤ 20*
Type	0 – 9
Amount	1 – 11
Channel Quality	*≤ 20*
Sinuosity	1 – 4
Development	1 – 7
Channelization	1 – 6
Stability	1 – 3
Riparian/Erosion	*≤ 10*
Width	0 – 4
Floodplain quality	0 – 3
Bank erosion	1 – 3
Pool/Riffle	*≤ 20*
Max depth	0 – 6
Current available	-2 – 4
Pool morphology	0 – 2
Riffle/run depth	0 – 4
Riffle substrate stability	0 – 2
Riffle embeddedness	-1 – 2
Gradient	*≤ 10*
Total Score	***≤ 100***

Source: Rankin[9]

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3. A CONCEPTUAL APPROACH FOR INTEGRATED WATERSHED MANAGEMENT

In Section 2.4 a rationale was presented for ecological risk assessment (ERA)-economic integration in watershed management: (a) that both risks and actions to reduce risk have an economic dimension, because they invoke preferences and trade-offs; (b) that technical information about risks, as is provided by ERA, is necessary for the formation of informed preferences; and (c) that the compartmentalization of disciplinary efforts leads to a poorer quality of analysis. It was recommended that whenever both ERA and economic analysis are needed to address a watershed management problem, they should be undertaken in an integrated fashion, which means that they should be mutually informed and fully coordinated. The goal of this chapter, then, is to develop a generalized, conceptual approach for achieving ERA-economic integration in a watershed management context. The conceptual approach has a similar form and purpose as existing frameworks developed by the U.S. Environmental Protection Agency (USEPA) such as the *Framework for Ecological Risk Assessment*[1] or the *Framework for Assessment of Ecological Benefits*.[2] This work draws from those and other frameworks, but the term framework is not used so as to emphasize that it is not intended to replace them.

This chapter first examines existing frameworks that have been used for watershed management, then considers some guiding principles, and finally presents a new conceptual approach that incorporates ERA into a well-integrated management process.

3.1 EXISTING FRAMEWORKS FOR WATERSHED MANAGEMENT

Various frameworks, emanating from the fields of risk assessment, environmental monitoring, project planning, environmental regulation and natural resource management, have been applied to watershed management processes, but none has addressed specifically the ERA-economic integration problem. Review of these frameworks reveals several characteristics by which they differ, which will be seen later to have bearing on the integration problem. The first of these is comprehensiveness with respect to the management process. Some frameworks address only *monitoring* or *assessment*, stopping short of decisions, whereas others are for *planning and management* as a whole, including decisions (and often, implementation, evaluation and adaptation). The second has to do with the intended use. Some can be termed *situational*, or responding to the advent of a problem or opportunity; others are for ongoing management and may be termed *regular*. The third characteristic is disciplinary breadth. Some frameworks are focused within the *natural sciences* whereas others emphasize both the *natural and social sciences*. The final characteristic is the degree to which the process is open to stakeholders, ranging from no explicit role to a role that entails negotiation rights. These four characteristics have been used to create an illustrative typology of some existing frameworks (Table 3-1). A discussion of each of these frameworks, in relation to the typology, is presented in Appendix 3-A.

3.2 GUIDING CONSIDERATIONS FOR AN INTERGRATED MANAGEMENT PROCESS

Given the existing frameworks, what considerations should guide the design (via borrowing and adaptation) of an approach for ERA-economic integration? According to USEPA's Science Advisory Board,[3] the processes used should have the following characteristics: they should be transparent (clearly understandable) to all parties; flexibly applied; dynamic (interconnected and iterative); open and cooperative; informed by many different sources and disciplines; and they should reflect holistic, systems thinking. Bellamy et al.[4] comment on the tendency for natural resource management efforts to fail to develop clear goals, achieve an integrated perspective, match actions to objectives, and evaluate outcomes

TABLE 3-1

Typology of frameworks that have been applied to the processes of watershed assessment and management.[a] Bold, bracketed numbers indicate degree of stakeholder integration in the process;[b] *italics* indicate an emphasis on the integration of natural and social sciences.[c]

	Monitoring and Assessment	**Planning and Management**
Situational: For project design or problem response	EMAP (Environmental Monitoring and Assessment Program) indicator design[5]**[0]** *DPSIR (Driving forces, Pressures, State, Impacts, Response) indicator framework design*[6]**[0]** Guidelines for ecological risk assessment[1]**[1]** *Framework for the economic assessment of ecological benefits*[2]**[1]**	Society for Environmental Toxicology and Chemistry's ecological risk management framework[7]**[1]** Framework for environmental health risk management[8]**[2]** *U.S. Army Corps of Engineers project planning*[9]**[2]** *World Commission on Dams planning and project development framework*[10]**[3]** USEPA's watershed project guidance[11]**[3]**
Regular: For ongoing management of watershed resources	Monitoring program with cyclical redesign[12] **[0]**	Clean Water Act watershed management cycle[13] **[2]** *U.S. Forest Service land and resource management planning framework*[14,15] **[2]**

[a] See Appendix 3-A for description of cited frameworks.
[b] Bold, bracketed numbers are further explained as follows:
[0] - No explicit stakeholder role: process may be amenable to stakeholder involvement, but such involvement is not described
[1] - Stakeholder-informed process: stakeholder involvement occurs primarily at the outset, as part of goal-setting
[2] - Stakeholder-engaging process: stakeholder involvement is sought throughout the process
[3] - Stakeholder-empowering process: process occurs at the initiative of stakeholders themselves; or framework deals explicitly with issues of "power" and assigns specific rights to stakeholders
[c] Integration of social sciences denotes the use of scientific methodologies, not stakeholder inclusion alone. It includes economics and the decision sciences.

They develop a broad set of criteria for evaluating efforts that have been implemented. These criteria are useful prospectively as well and are presented here as relevant to the development of an integrated process. They state that an effective process

> (a) addresses evaluation from a systems perspective, (b) links objective to consequence, (c) considers the fundamental assumptions and hypotheses that underpin core policy or program objectives, (d) is grounded in the natural resource, policy/institutional, economic, socio-cultural and technological contexts of implementation in practice, (e) establishes practical and valid evaluation criteria by which change can be monitored and assessed, (f) involves methodological pluralism including both quantitative and qualitative methods to ensure rigor and comprehensiveness in assessment, and (g) integrates different disciplinary perspectives (i.e. social, economic, environmental, policy and technological).

Based on these ideas, issues raised in Chapter 2 and the examination of other frameworks, a set of considerations that address watershed management generally, and are also specific to the ERA-economic integration problem, are listed in Table 3-2. These considerations, and the design elements resulting from each, are summarized below.

As was emphasized in Chapter 2, ERA has unique value as an ecologically informed process that conceptually defines the ecological system at hand and the anthropogenic forces acting upon it and that progresses, in structured and logical fashion, from ecosystem management goals to the characterization of risks affecting those goals. An integrated framework should retain the processes composing the analytic core of ERA, and the essentially scientific character of the analysis should not be compromised. At the same time, in order to secure broad participation leading to robust solutions, there must be sensitivity to the critiques of ERA discussed in Section 2.1.2, particularly that ERA can be too narrowly focused: bearing the mantle of "science" yet serving particular interests[16] or lacking a clear link to management efforts.[17] These criticisms may be answered by an approach that emphasizes the comparative

TABLE 3-2 Important considerations in framework design, and resulting design elements		
Consideration	Specific points	Framework design element
Unique value of ERA	• ecologically informed, biophysical in nature • structured, deductive process proceeding from goal --> objectives --> hypotheses --> analyses --> risk characterization	• scientific character of analysis is not compromised • retain core of ERA process
Sensitivity to critiques of ERA	• some stakeholders may perceive overall process as unfair • assess broad range of alternatives (not constrained set) • acknowledge limits of science throughout assessment	• acknowledge potential "winners" and "losers;" extend negotiating rights • comparative assessment of alternatives • "deliberation" by "extended peer community" throughout process
Key aspects of economic thought	• individual preferences and trade-offs are essence of value • citizen sovereignty (as constrained by mandate of representative government)	• comparative assessment of alternatives • stakeholders in process; analysis of preferences • risk communication is inside process
Methodological pluralism	• neither ERA nor economic analysis ascendent • both deliberative (constructive) and logico-deductive processes can inform decisions	• extended peer community • decision is based on input from multiple disciplines
Importance of adaptive management	• costs and uncertainties often high; politics may not bear full implementation • assume incremental, negotiated decisions; include analysts here and in subsequent steps	• negotiation part of decision process • adaptive management integral, not accessory
Linkage of situational and regular management processes	• both types of processes are needed • should be mutually supporting	• linked cycles

assessment of a range of management alternatives; that identifies stakeholder groups that are likely to bear the respective risks and benefits of the alternatives; and that sees negotiation among these groups as legitimate. Where uncertainties and decision stakes are high, the approach should acknowledge the limits of science by accommodating "deliberation"[18] by "extended peer communities"[19] throughout the process. Scheraga and Furlow[20] coined the term "policy-focused assessment" to describe a scientific process that is constantly engaged with stakeholders and decision-makers so that the results will be relevant to policy.

The incorporation of economics into the process implies there will be an increased emphasis on the measurement of individual preferences, expressed as the willingness to make trade-offs. This dynamic reaffirms the importance of comparative assessment of alternatives. It also implies that risk communication, necessary for informing preferences, is an essential component of an integrated process (whereas it is accessory to ERA); and that stakeholder preferences will be analyzed in some form.[a]

Methodological pluralism[21] is a relevant goal because the salient attributes of environmental management problems are not adequately modeled by any single disciplinary paradigm. The extended peer community should include multiple disciplines;[21] both qualitative and quantitative data collection methods may be needed; and deliberative as well as deductive processes may be relevant. In the decision-making phase it may not be possible to reduce all relevant factors to a single dimension: multiple objectives may need to be treated.

Adaptive management has been described as a "learn-by-doing" approach to decision-making, in which both goals and approaches are subject to revision over time.[22] When

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[a] For guides on sharing environmental information with the public, refer to USEPA;[44,45] for useful information on terminology for communicating ecological concepts, see Schiller et al.[27] and Norton.[46]

the process is applied to the implementation of a plan or policy, rather than the ongoing management of a resource, the term "adaptive implementation" may be used.[23] Analytical frameworks often treat adaptive implementation as an accessory process – a post-analytic feedback loop which acknowledges that uncertainty and complexity may prevent us from precisely hitting the target on the first try. Experience, however, suggests something more. Where costs of remediation or restoration are high, the political will to take fully responsive actions may be lacking, even where scientific knowledge is relatively adequate. Interested parties might first negotiate a less costly, interim decision. Adaptive implementation could then constitute an indispensable learning process through which a community gradually acquires willingness to take more vigorous steps. As Holling et al.[24] put it, "managers as well as scientists learn from change," and the same can be said for other stakeholders. If so, it would be a mistake to view negotiation as a purely nonscientific process taking place after the specialists have "had their say." Rather, technical specialists should participate in the design of an incremental process that yields information and employs evaluation criteria at each step. They should be expected to play a supporting role during negotiations and to be actively engaged through the adaptive implementation process.

Finally, because environmental management entails both regular and situational processes, it may be important to examine how the problem-oriented process of ecological-economic analysis, decision-making and adaptive implementation that is being developed herein can best interact with ongoing resource management requirements.

3.3 DIAGRAMING AN INTEGRATED MANAGEMENT PROCESS

Figure 3-1 diagrams a conceptual approach that addresses each of the guiding considerations listed in Table 3-2 and, in so doing, responds to each of the SAB and Bellamy

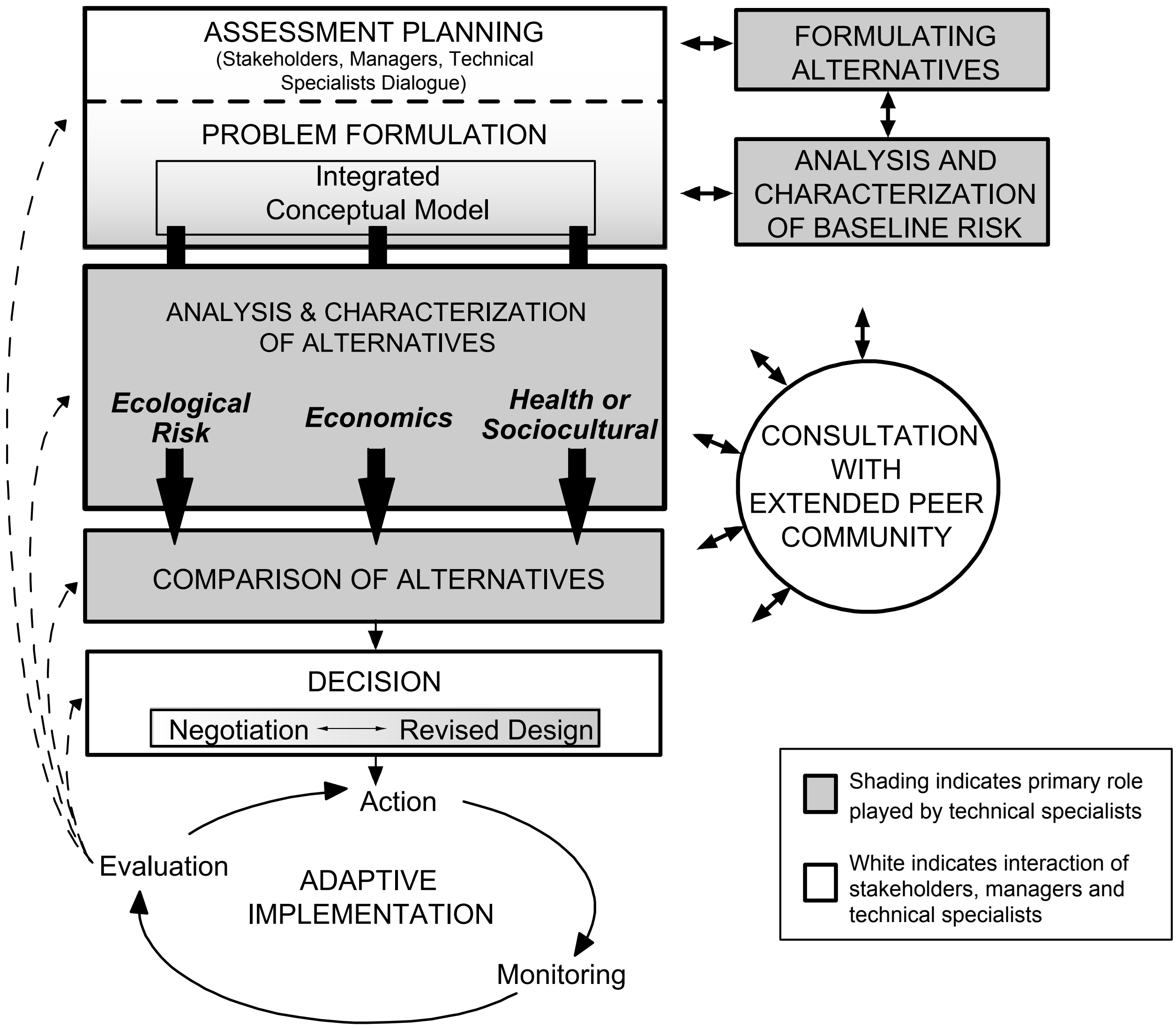

FIGURE 3-1

A conceptual approach for the integration of ecological risk assessment and economic analysis in watershed management criteria. The major components are discussed in the succeeding sections. In many respects the approach is similar to the ERA *Framework*. However, ERA only estimates the likelihood of adverse ecological effects, and it assumes that economic analysis, if needed, will be able to use the assessment results. This approach modifies the ERA *Framework* at every stage of risk assessment, beginning with the planning process, to ensure compatibility. In so doing, however, the core scientific character of ERA is not compromised. The scope of planning and problem formulation are broadened but the key steps of articulating ecological values, goals, objectives, and endpoints are still carried out. Analysis and characterization of ecological risks is carried out in a scientific manner as part of the analysis of management alternatives and sometimes also as part of an assessment of baseline risks.

This conceptual approach would be placed in the upper right cell of the typology presented earlier (Table 3-1); that is, it is a situational process, triggered by need rather than ongoing. However, it includes an adaptive implementation phase, which may continue, and it can be linked to or used within an ongoing watershed management cycle. It is a planning and management approach that includes decision-making and implementation; it is not limited to providing information for decision-support. It generally assumes that stakeholders and decision-makers will be involved in the initial stages and will remain engaged at some level throughout the process, such as through consultations with an extended peer community, but that analysis and characterization will be conducted by technical specialists. Depending on the decision context, stakeholders may be empowered to participate in or to make decisions (i.e., it would be scored as **[2]** or **[3]** in Table 3-1). Each of these aspects is further discussed in the following sections.

The sequence of discussion is not necessarily that in which the process will occur. The process may begin with assessment planning, initiated because a problem or opportunity has been recognized. On the other hand, a proposal for one or more actions may have been formulated that now requires full evaluation, or a study of baseline risks (i.e., present and future risks, if no new action is taken) may have been conducted that demonstrates a need for actions to be formulated and comparatively assessed. A separate step for the study of baseline risks is not needed at all if the analysis of alternatives includes the no-action alternative. However, assessment planning, problem formulation and formulation of alternatives all should be completed prior to the assessment of alternatives and subsequent steps (although the reiteration of these steps may be necessitated by later findings, or by intervening events).

3.3.1 Assessment planning

Assessment planning is analogous to "planning" in ERA and to "identifying problems and opportunities" in the U.S. Army Corps of Engineers (USACE) project planning process; it is here termed assessment planning to distinguish it from the more encompassing terms "project planning," used by USACE, and "resource management planning" used by the U.S. Forest Service (see Appendix 3-A). It is a stage that emphasizes discussions among analysts of multiple disciplines (i.e., ecological, economic and others as needed), risk managers and, where appropriate, stakeholders about values and goals. It is conducted as described in Section 2.1.1.1, except on three major points. First, the identification of the decision context is somewhat expanded. Besides identifying the decisions to be made and determining their context, assessment planners must also determine who has the authority to make the decisions and what criteria they expect to use. These are critical factors for the characterization and comparison of alternatives; analysts need to know how the decision-makers view the decision situation so their comparisons comprise all the needed elements. For example, decision-makers may be specifically constrained to consider, or not to consider, particular factors such as cost, equity or threatened and endangered species, or to prioritize some factors vis-à-vis others.

Second, the scientific disciplines needed to address all important dimensions of the problem should be represented in assessment planning. Besides ecology and economics, which are the focus of this document, the watershed management problem may have implications for human health, requiring the involvement of health risk assessors. In addition, sociocultural issues such as environmental justice concerns or threats to cultural artifacts could require the parallel involvement of additional disciplines (geography, cultural anthropology, archeology, etc.), here and throughout the assessment process. These various analysts should help decision-makers elucidate their time horizon of concern. Decisions have both short- and long-term consequences, and ecological and economic time frames of analysis will need to acknowledge the time horizons of the relevant processes involved, the decision-makers and the other disciplines.

Third, not only must interested and affected parties be identified, but the ways in which they may be benefited or harmed by the alternatives under consideration should be indicated because, depending on the legal context, it may be necessary or advisable to accord them negotiating rights, or to address compensation issues, in the decision process. This information will also be useful if the negotiation process is to be modeled (e.g., using game theoretic techniques, see Section 2.2.5).

3.3.2 Problem formulation

In the ERA *Guidelines*,[1] problem formulation is a scientific process that is kept separate from planning (see Figure 2-1, and refer to the discussion of problem formulation in Section

2.1.1.2). As shown in Figure 3-1, however, it is separated from assessment planning by a dashed line to indicate the tendency for these two steps to be closely associated in practice. For example, conceptual models produced in problem formulation diagrammatically illustrate for stakeholders and decision-makers the complex causes, nature and ramifications of ecological problems in watersheds,[25] as is necessary for assessment planning.

The distinction between these two steps is further reduced here because of the need to broaden conceptual models and assessment endpoints to include socioeconomic as well as ecological impacts – an exercise that is likely to rely on repeated discussions with interested and affected parties. In ERA, risk hypotheses, which are proposed explanations of relationships between sources, stressors, exposure pathways, receptors and ecological effects, are the basis of conceptual models (see Section 2.1.1.2). To include socioeconomic impacts, risk hypotheses must be extended to include the changes in ecosystem services (see Table 2-1) that will be associated with the changes in those endpoints. Finally, since the evaluation of alternatives is also required for an integrated assessment, risk management hypotheses are needed as well; these are proposed explanations of how management alternatives will affect sources, exposures, effects and services.

Section 2.1 used the example of the decline of a hypothetical reservoir fishery to illustrate the components of ERA. Section 2.1.1.2 listed population size, mean individual size and recruitment of popular angling species as appropriate ecological assessment endpoints, and it stated that conceptual models should diagram the ecological processes whereby the stressors suspected of causing the decline, in this case agricultural pesticides and municipal and agricultural nutrients, were thought to exert effects. Continuing that example, the integration of economics at the problem formulation stage would require adding management alternatives to the conceptual model. In this example, suppose that a baseline risk assessment (see Section 3.3.3) had identified nutrient loadings to the reservoir as the actual cause of the decline, and that risk management alternatives to be studied (see Section 3.3.4) included restricting further sewerage connections to the municipal treatment plant, upgrading the treatment plant, instituting an incentives program for riparian zone restoration, and conducting an outreach program to encourage conservation tillage. Extending the conceptual model would require adding each of these alternatives to the diagram and illustrating their expected effects on the ecological processes relevant to the endpoints. Additional effects that might have ecological relevance would be diagrammed as well, such as important beneficial or detrimental effects on species that were not the original subject of the assessment. These might require defining additional ecological assessment endpoints.

Economic effects of the alternatives must also be added to the model. Since the ecological assessment endpoints in this example (fish species, population, size, etc.) are not directly valued, the link to ecosystem services such as fishing success must be included in the diagram, and assessment endpoints corresponding to the service changes (for example, value to recreational users) must be added. Other economic effect pathways, such as the effects of plant upgrade costs or land use changes on the local economy, also need to be included. Finally, other kinds of changes expected to result from the alternatives, such as changes in human health or quality of life, should also be indicated. Complete risk management hypotheses will consist of a causal chain that extends from a given management alternative to each of the applicable ecological and economic assessment endpoints.

The analysis plan, which is the final product of problem formulation, must include procedures for evaluating the risk management hypotheses, including the efficacy of proposed management actions and the relationship between ecological responses and ecosystem services. The plan must include quantification of the spatial and temporal extent of endpoint changes.[26] (In the reservoir example, ecosystem service improvements resulting from a management action would depend on the size of the area over which the fishery was improved and the time required to effect the improvement.) The plan must also include proposed methods for the comparison of alternatives that closely reflect the needs of decision-makers, as determined during assessment planning (see Section 3.3.7 for further discussion of comparison). Finally, the analysis plan and other products of problem formulation (assessment endpoints and conceptual models) must be verified with managers and stakeholders as being not only technically accurate but well-targeted to the most important concerns. If members of these groups have been engaged throughout assessment planning and problem formulation, they may have acquired in the process sufficient technical knowledge to understand these products. If not, or if the economic methods to be used later will require surveys of a broader audience or the general public, then careful work will have to be done at this stage to build a risk communication capability. Steps may include developing common-language terminology to express key ecological concepts,[27] and using focus groups to refine this lexicon and verify assumptions about the values held by the public or stakeholder groups.

3.3.3 Analysis and characterization of baseline risk

If preexisting information is not sufficient, a separate study of baseline risks may be conducted prior to the formulation of alternatives. Although definitions can vary slightly, baseline risks are defined as the present and future risks to ecosystems or human health that would occur if no new action is taken.[28] Baseline risk assessment is a formal part of environmental impact assessments conducted under the National Environmental Policy Act

(NEPA) and site characterizations conducted under the Comprehensive Environmental Response, Compensation and Liability Act ("Superfund"). Since NEPA requirements are invoked only when an action is proposed, the action alternative and no-action alternative are assessed in the same stage of environmental impact assessment, and baseline assessment as a separate step is not needed. Under Superfund, on the other hand, baseline assessment is needed to characterize the risks prior to remedial action design. In watershed management, a separate baseline assessment as shown in Figure 3-1 may be required if the kind of management action needed, or the need for any action at all, is unclear.

Characterizing baseline risks may also require characterization of harms that have already occurred. Risks to socioeconomic well-being may also form part of this analysis, but these risks are more easily addressed in comparative than absolute terms and are therefore likely to receive limited attention at this stage. Methods for analysis and characterization of ecological risk were discussed in Section 2.1.1; methods for the assessment of health risks are presented elsewhere.[8,29] Determining the magnitude and severity of ecological or health effects helps determine the need for management actions. Determining causality and pathways of exposure provides information useful in the design of management alternatives. Developing models of exposure and response, and risk characterization approaches, establishes the methods that will be used in the comparative analysis of management alternatives.

The generation of exposure scenarios may be an important part of baseline risk assessment. Scenarios are often used to describe alternative circumstances for which risk will be estimated. In some instances they help describe the range of the expected exposure conditions; for example, an assessment of pesticide impacts on watershed resources may require setting up a range of use scenarios to cover the different types of practices actually occurring in the watershed. Exposures resulting from all scenarios would then be used in the full characterization of baseline risk. In other cases, scenarios result from alternative assumptions about an unknown future; for example, alternative CO_2 emission assumptions and global climate models are being used to establish alternative future climate scenarios for watershed risk assessment.[30] These scenarios are part of baseline assessment if they do not correspond to designed policies or alternative management actions but rather form a positive basis for design of management actions. On the other hand, some future scenarios are explicitly policy-based. For example, Coiner et al.[31] developed future scenarios for the Walnut Creek watershed of Iowa based on alternative policies that respectively prioritized agricultural production, water quality and biodiversity; and Hulse et al.[32] developed scenarios for the Muddy Creek watershed of Oregon reflecting different policies with respect to development density and conservation. Policy-based future scenarios, which enable a normative comparison of policy outcomes, would be developed as part of the next stage, "Formulation of Alternatives."

3.3.4 Formulation of alternatives

This phase entails the development of alternative action plans for achieving the watershed management objectives. Depending on the nature of watershed problems and the management goals, there is a wide array of management actions that may be considered at this step (Table 3-3). The planning process may include engineering design or policy development; the discussion of specific techniques is beyond the scope of this report. Details of processes that can be used for developing alternative plans are presented elsewhere.[9,13,33-35] [13,33,34,34,34,35,35,35]While actions to reduce ecosystem risks are emphasized in this report, actions designed to reduce human health risks or improve socioeconomic well-being may cause ecological changes and therefore may also need to be evaluated according to the procedures in this chapter.

TABLE 3-3 Categories (and some examples) of watershed management measures
Control of point sources (source reduction, waste recycling, waste pretreatment, or improvement of waste treatment infrastructure)
Control of urban or agricultural nonpoint sources (land use changes, runoff detention structures, improved waste management, educational outreach programs)
Contaminant remediation (chemical spill cleanup, acid mine drainage treatment)
Stream channel and riparian restoration (tree planting, instream structures)
Species management (habitat creation, control of nonnatives, reintroductions)
Water resource development (irrigation, hydropower, recreation)
Improvement of other use values (access)
Strategies for adaptation to global change (land use changes to accommodate sea-level rise)

To avoid bias toward preselected solutions, planning objectives and constraints should be clearly established in advance,[9] and a broad range of alternatives should be examined (see Section 2.1.2). A given alternative should comprise not just the design of management actions such as those listed in Table 3-3; long term success depends on establishing a planned system that also includes implementation tools (such as permits, incentives and information) and institutional and organizational arrangements (such as extension services).[35]

3.3.5 Consultation with extended peer community

Funtowicz and Ravetz[19] describe "extended peer communities" as including scientists outside the specific discipline or practice at hand, and others lacking formal knowledge but possessing practical, including local, knowledge (see Section 2.1.2). The term used here includes interested and affected parties and decision-makers, in addition to scientific peers. "Consultation" does not apply to the assessment planning phase, where interested and affected parties are already an integral part of the process. It applies rather to components such as analysis and characterization that are explicitly scientific. "Consultation" recognizes on the one hand that these steps must be carried out by analysts with specialized knowledge, and on the other that risk assessment often requires judgments that go beyond strict inference and are therefore susceptible to bias. Consultation is a process in which technical information from the assessment is discussed with the extended peer community for purposes of (a) identifying issues or deficiencies in the assessment and (b) keeping interested and affected parties engaged during what can be a lengthy process. It is equivalent to the term "deliberation" as used by NRC.[18]

3.3.6 Analysis and characterization of alternatives

In this stage the alternatives are assessed from the perspective of various disciplines including ERA, economics and possibly others such as human health risk or sociocultural assessment, depending on the situation. In the diagram (Figure 3-1), the disciplines are shown as jointly conducted, indicating at least an exchange of information and at best an integrated analytic approach. However, it is by no means a requirement that the disciplines depart from their characteristic approaches, as long as they are mutually informed. Since ecological and economic time frames of analysis may differ, the time frame for each should be made explicit.

Analysis of alternatives is guided by the risk management hypotheses, indicating which exposures and responses are likely to be affected by risk management. Those not expected to be affected remain part of the baseline risk but are not included in the alternatives analysis. The ecological risk component estimates the changes in exposure profiles likely to result from each management alternative. Where management alternatives create new exposures, i.e., to stressors that were not originally present (such as to sediments from project construction or to pesticides used to control invasive species), additional exposure profiles and exposure-response relationships beyond those of the baseline assessment must be developed.

Ecological risk characterization describes probabilities, magnitudes and severities of effects on ecological assessment endpoints. These should be described both in absolute terms and as changes with respect to baseline. Uncertainties in the effect estimates must be characterized as well, and the uncertainties, as well as the other parameters, must be carried forward into the economic analysis.[26]

The economic component analyzes costs (including financial and opportunity costs) and benefits associated with the management alternatives. This includes, to the extent practicable, the changes (with respect to the no-action baseline) in ecosystem services that are associated with changes in the ecological assessment endpoints. This especially includes services that can be quantified objectively, such as biophysical services (e.g., the production of food, fiber or other goods, and regeneration and stabilization processes) and services that are quantifiable by revealed preference methods (e.g., many forms of recreation). It may also include life-fulfilling functions (including functions corresponding to non-use values), if these can be quantified by benefit transfer methods. The use of stated-preference or other subjective methods to quantify these services is not ruled out at this stage, but for pragmatic reasons such efforts may best be carried out as part of the subsequent, comparison phase. For example, if a stated preference questionnaire were to be designed and administered, it may be possible, and therefore cost-effective, to do so in such a way as to affect a multifactoral comparison, as described below.

3.3.7 Comparison of alternatives

This step is included in the conceptual approach based on the assumption that not all factors important for decision-making can be objectively reduced to a single vector and that the comparison step itself therefore is both subjective and nontrivial. Even if net economic benefit to society, as determined by CBA, is an important criterion, there will usually be other ecological, moral, political or legal factors that it cannot adequately encompass. Comparison is the step in which these various factors are arrayed in terms as amenable as possible to those of the legitimate decision-maker, be it an agency official, the collective of residents of a jurisdiction, or individual landowners. Any process used to assign subjective weights to the factors, or to enable individuals or groups to systematically compare the alternatives (based on information about these factors and their subjective judgment) is considered to be part of the comparison phase. Methods may include stated preference analyses (appropriate for large groups of individuals) or decision-analytic approaches in which factors are weighted by technical experts, or by representatives of interested and affected parties, acting either as individuals or within consensus-seeking groups (see Morgan[36] for a useful summary of non-monetary, multicriteria evaluation methods). On the other hand, if the ultimate decision will be reached by negotiation among parties with divergent interests, the comparison methods used might seek to identify the alternative that the parties believe is the best they can hope to obtain, rather than the one with optimal overall utility. The comparison process is carried out according to agreements made during the assessment planning phase (in which the decision context, including decision-makers and decision factors, was described) and the problem formulation phase (in which the comparison methods to be used were verified).

3.3.8 Decision

Because environmental management problems in watersheds usually are multidimensional, it is unlikely that a problem can be solved based on the actions or authority of any single entity. Therefore, the decision process is likely to involve multiple parties. In spite of the findings of the analysis and characterization of alternatives, or because of the associated uncertainties, the parties may hold divergent beliefs about expected outcomes of a given alternative, or even if they agree on technical issues they may have divergent incentives or expectations regarding compensation. They may also have divergent interpretations of legal constraints on the decision process. Therefore, a decision may entail less a consensus selection among the alternatives than a negotiated redesign. Where implementation cost is a predominant factor, negotiation may entail scaling back on a design or agreeing to a provisional schedule of incremental implementation, conditioned on verification that performance criteria are being met. Technical specialists therefore may be called on to assist the negotiation process.

3.3.9 Adaptive implementation

Because achieving agreement can be difficult, a provision for adaptive implementation may therefore be indispensable to reaching a decision: it can provide a middle-ground approach that satisfies no one but provides a respite until confirmatory data are available. However, a flexible or incremental approach does not constitute adaptive management unless several criteria are met. Holling et al.[24] recommended that experimental perturbations be designed to evaluate specific questions. Walters[37] emphasized that perturbations need to be great enough to probe system responses across domains of interest; cautious incrementation may not produce any usable information. The National Research Council (NRC)[22] stated that adaptive management must not only generate useful information but must specify the mechanisms by which the information will be translated into policy and program redesign. Depending on the findings and the nature of the agreement, evaluation of the data could lead to further action or could trigger renewed negotiation; it could also invalidate certain assumptions of planning, problem formulation or analysis, indicating that earlier stages of the process need to be reiterated. The possibility of revisiting earlier steps in the assessment as more information is learned is indicated by broken lines in Figure 3-1.

3.3.10 Linkage to regular management cycles

The process described here for integrated assessment is situational; i.e., it should not be thought of as a cyclical process that can never be completed. By contrast, resource management is ongoing, and the two processes can be mutually supportive. For example, the rotating basin approach to CWA management (see Appendix 3-A) identifies priorities and needed actions, which may call for a detailed, integrated assessment in situations where needed actions are unclear or where regulatory approaches are insufficient. Stakeholder processes that may have been established as part of that cycle can be drawn upon for the integrated assessment. The rotating basin approach also establishes a long-term water-quality and biological monitoring data base that can establish temporal trends and correlations in stressors and biological response that can be useful in establishing causation, exposure profiles and stress-response relationships. The management alternatives to be considered in the integrated analysis can include (among other measures) regulatory and incentive mechanisms provided for under the CWA, to be implemented and monitored as part of the regular management cycle. Similarly, some watershed resources (e.g., forest resources) are adaptively managed in an ongoing fashion (see Appendix 3-A, Figure 3-A-5). Integrated assessments can link effectively to an adaptive management cycle.

3.4 EXAMPLES OF ANALYSIS AND CHARACTERIZATION FOLLOWED BY COMPARISON OF ALTERNATIVES

Planning and problem formulation (together with baseline ERA and formulation of alternatives) lay the groundwork for a successful integrated analysis, but the technical aspects of integration are encountered in the analysis and characterization of alternatives and in the comparison step that follows (Figure 3-1). Because there are a variety of ecological and economic analytic tools that could be applied in these stages, the specific elements of these steps will also vary. This section provides examples to illustrate how ecological and economic techniques might interact.

3.4.1. Example 1: Cost-benefit analysis of all changes that can be monetized, with qualitative consideration of other changes

Cost-benefit analysis (CBA, see Section 2.2.3) is commonly used where decision-makers are concerned about the net economic benefit to society of a given action (that is, to determine whether economic efficiency is increased).[38] As discussed in Section 2.3.2, CBA is required for certain federal actions. In an integrated assessment where changes in economic efficiency will be a key factor in the decision, the process may occur as diagramed in Figure 3-2. For each management alternative, ecologists would quantify the changes expected in each ecological assessment endpoint. Changes that could not be quantified would be characterized qualitatively.

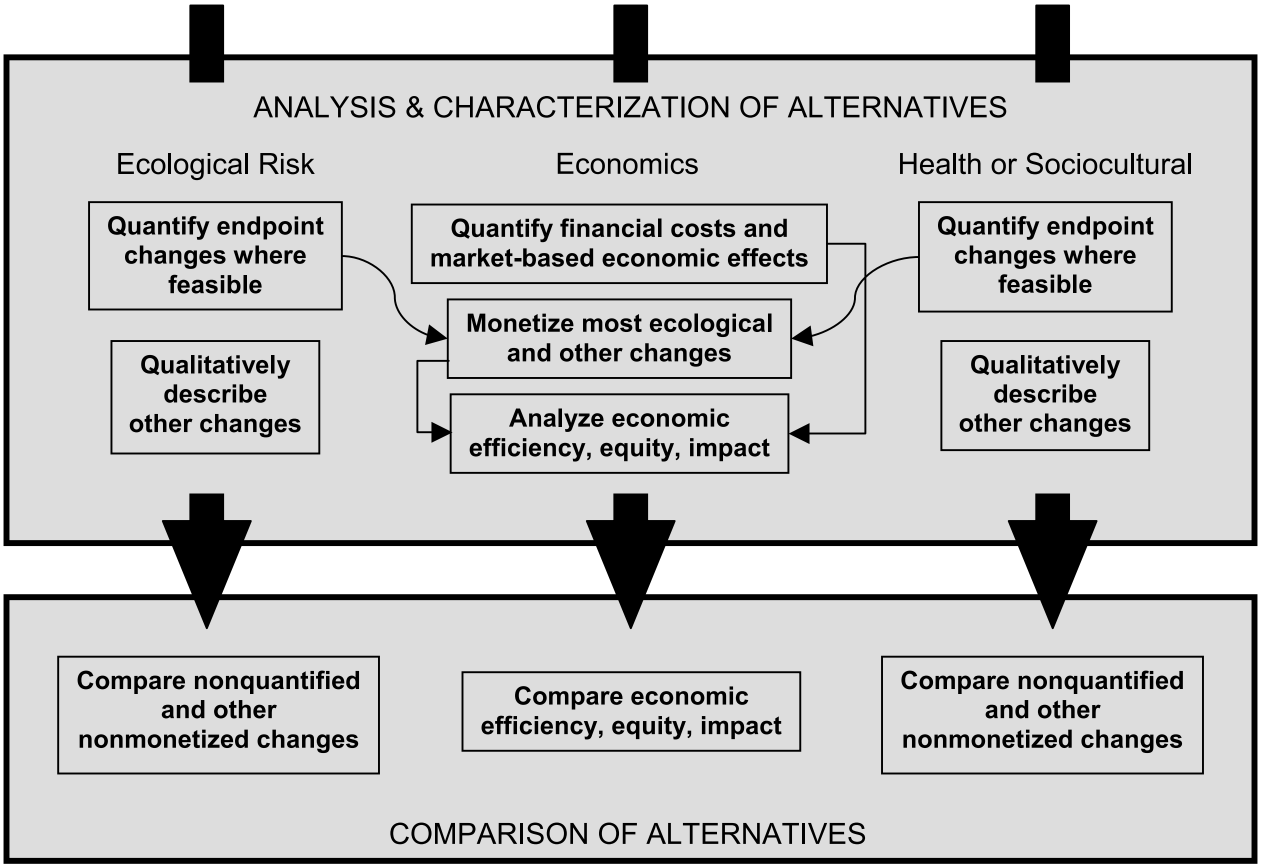

FIGURE 3-2

Analysis and characterization of alternatives, followed by their comparison, example 1: CBA of all changes that can be monetized, with qualitative consideration of other changes.

Other analysts might examine quantitative or qualitative effects on health or quality of life, as needed. Economists would look first at the financial costs of the alternatives and any effects that could be determined from markets (for example, opportunity costs of land taken out of agricultural production). Economists would then seek to monetize the effects estimated from the ecological or other analyses, using revealed preference or benefit transfer methods wherever possible (see Section 2.2.2 and especially Table 2-2). Due to their required time and cost, stated-preference techniques would be used only if other methods were unsatisfactory (that is, if nonuse values are important and/or reliable studies from similar settings are lacking). Based on this information, economists would analyze economic efficiency, equity and impacts (Section 2.2.4). This information, and information about effects that either could not be quantified or could not be monetized, would be carried forward into the comparison step.

3.4.2 Example 2: Use of stated preference techniques to effect integration of ecological, economic and other factors

In the example above, stated preference methods, if used at all, would monetize the ecological changes associated with one or more management alternatives. Figure 3-3 diagrams the use of stated preference methods to achieve a more broadly-based comparison, such as one that includes the ecological, health, quality-of-life, equity, and impact dimensions of a choice. For example, this could be accomplished using a contingent valuation method (CVM, see Appendix 2-A) survey that explains the effects of the management alternatives (i.e., that "frames" the alternatives) in each of those dimensions before asking individuals about their willingness to pay (WTP) or to accept (WTA) (see Section 2.2.2). To design such a survey, each of those dimensions would first need to be analyzed and characterized, with all effects quantified to the extent possible. The technical findings would then need to be refined (such as through the use of focus groups) into a format that highlighted only the most important factors and used

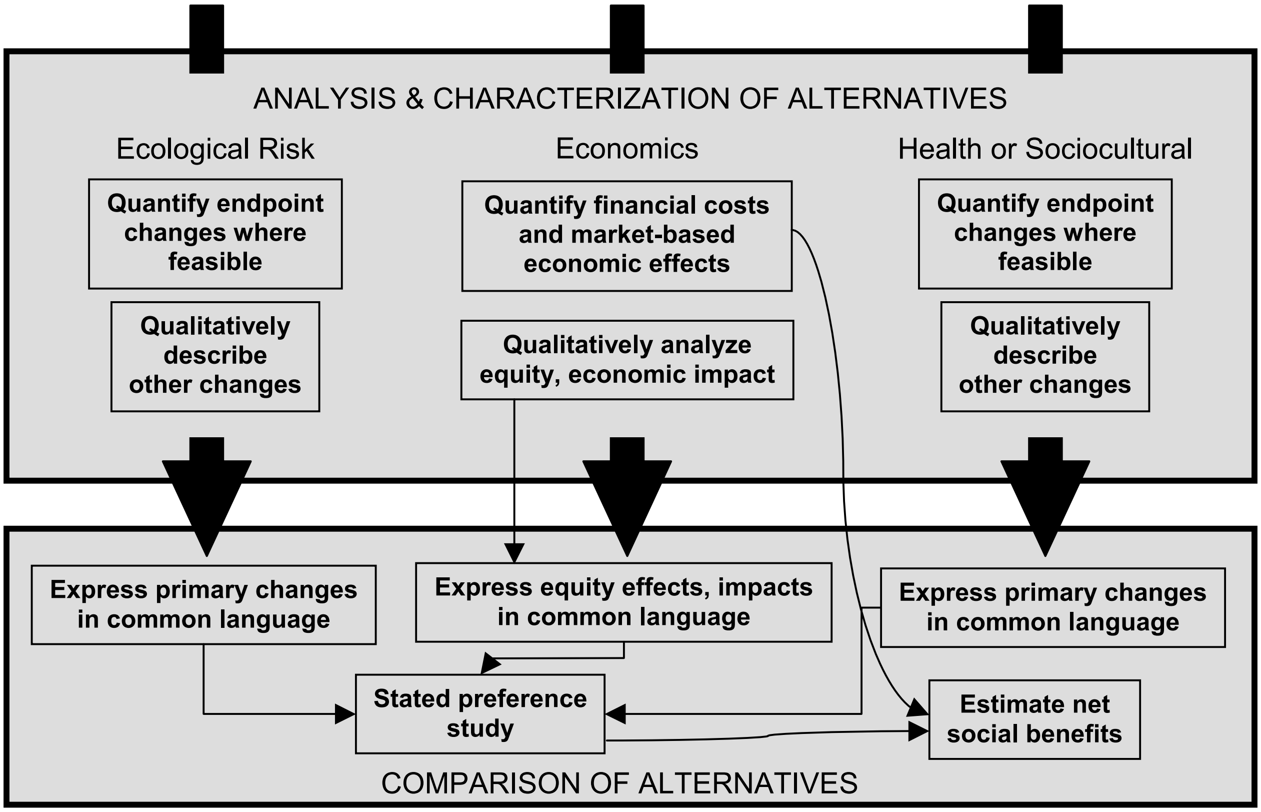

FIGURE 3-3

Analysis and characterization of alternatives, followed by their comparison, example 2: use of stated preference techniques to effect integration of ecological, economic and other factors.

commonly understood language.[27] A broadly-framed CVM approach that was similar to this in certain respects was employed in the Big Darby Creek watershed case study presented in Chapter 4.

A broad comparison could also be accomplished using a choice modeling method such as conjoint analysis (CA, see Appendix 2-A). In this approach, focus groups would again be used to identify the most important factors across those dimensions, and to establish common terminology. Survey design would entail transforming those dimensions into choice attributes, so that respondents' choices would reveal how the various dimensions contributed to WTP or WTA. A method of this type was used in the Clinch Valley case study presented in Chapter 5.

3.4.3 Example 3: Use of linked ecological and economic models to dynamically simulate system feedbacks and iteratively revise management alternatives

A disadvantage of sequentially integrated assessments, in which ecological changes are estimated and then economically evaluated, is that there is no opportunity to simulate dynamic interaction between economic and ecological processes.[39,40] In cases where the economic effects of changes in ecosystem quality (such as effects on housing, recreational or agricultural values[41,42]) will have an important influence on land use decisions and ecosystem quality, an integrated system that models these feedbacks may enable a better understanding of the behavior of the real systems. In Figure 3-4, models of the ecological processes affecting the assessment endpoints are linked to a regional economic model in a manner that allows parameter feedbacks over time. Once such a modeling system is established, management alternatives can be simulated and iteratively revised to optimize their design according to a variety of criteria, such as cost-effectiveness, equity and ecological risk. The example pictured in Figure 3-4 arbitrarily assumes a case where ecological and economic models are linked and that other effects (e.g., on health or quality of life) are estimated using other methods. The example further assumes that it

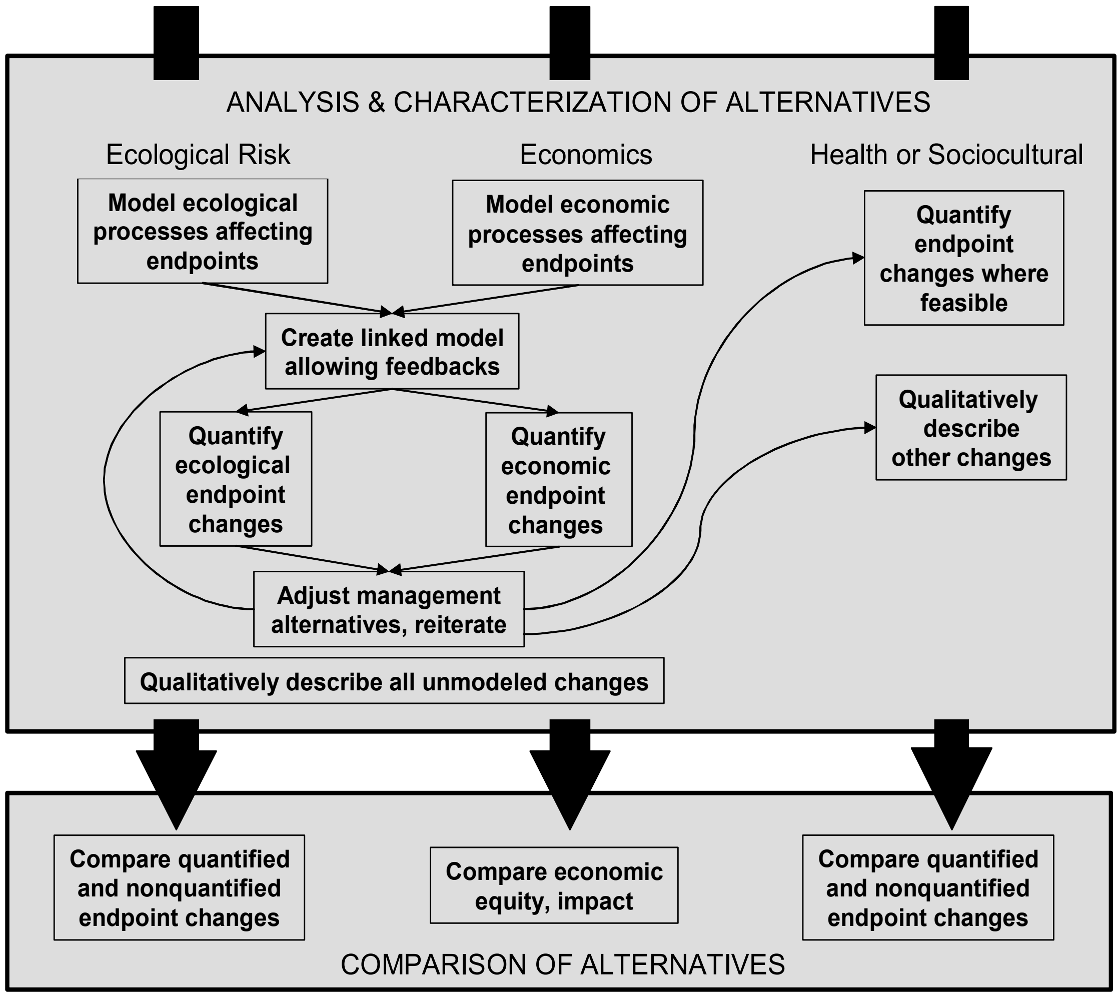

FIGURE 3-4

Analysis and characterization of alternatives, followed by their comparison, example 3: use of linked ecological and economic models to dynamically simulate system feedbacks and iteratively revise management alternatives.

may be difficult to estimate net social benefit from such a modeling approach, since WTP or WTA for nonuse values is not estimated, although in theory an appropriate benefit transfer module could be added to the model. In the comparison step, the modeling results for the various management alternatives and/or for different optimization criteria could be described, along with qualitative discussion of any effects that could not be quantified by the modeling effort

3.5 CONCLUSION

This conceptual approach does not represent a fundamental departure from existing practice. Its steps correspond in large part to those of other frameworks (Table 3-4); they differ as needed to emphasize the ERA-economic integration problem. However, the incorporation of multiple disciplines into an integrated assessment process may create significant challenges of communication, coordination and funding. Therefore the use of this approach is not appropriate in all instances where ERA alone is called for. However, if decisions need to be informed on the basis of both ecological risks and economics, an integrated approach, while more demanding, is more likely to provide coherent information.

This conceptual approach is used in the following chapters as a vantage point from which to analyze a set of case studies. As was mentioned in the previous chapter, the case studies that will be presented in Chapters 4-6 were undertaken with a number of constraints. In each case, the involvement of economists came well after ERA had been initiated, and in one case the ERA was never completed. Furthermore, the scope of these studies did not encompass the full span of management activities, from assessment planning to adaptive implementation. Nonetheless, the conceptual approach helps to illustrate how the methodological advances and insights from each

TABLE 3-4

Rough correspondence between the components of the conceptual approach for ERA-economic integration and other selected watershed management frameworks[a]

<table>
<tr><th rowspan="2">Component of Conceptual Approach for ERA-EA Integration (Figure 3-1)</th><th colspan="4">Corresponding Component</th></tr>
<tr><th>Framework for Ecological Risk Assessment [1]</th><th>Framework for Integrated Environmental Decision Making [3]</th><th>Watershed Management Model [43]</th><th>U.S. ACE Six-Step Planning Process</th></tr>
<tr><td>Assessment Planning</td><td>Planning</td><td rowspan="2">Phase I: Problem Formulation</td><td rowspan="4">Phase I: Assessment/Problem Identification</td><td rowspan="2">Identifying Problems and Opportunities</td></tr>
<tr><td>Problem Formulation</td><td>Problem Formulation</td></tr>
<tr><td rowspan="2">Analysis and Characterization of Baseline Risk</td><td>Analysis</td><td rowspan="7">Phase II: Analysis and Decision Making</td><td rowspan="2">Inventorying and Forecasting Conditions</td></tr>
<tr><td>Risk Characterization</td></tr>
<tr><td>Formulation of Alternatives</td><td>NA[b]</td><td rowspan="5">Phase II: Planning</td><td>Formulating Alternative Plans</td></tr>
<tr><td rowspan="2">Analysis and Characterization of Alternatives</td><td>Analysis (reiteration)</td><td rowspan="2">Evaluating Alternative Plans</td></tr>
<tr><td rowspan="2">Risk Characterization (reiteration)</td></tr>
<tr><td>Comparison of Alternatives</td><td>Comparing Alternative Plans</td></tr>
<tr><td>Decision</td><td>NA</td><td>Selecting a Plan</td></tr>
<tr><td rowspan="2">Adaptive Implementation</td><td rowspan="2">NA</td><td rowspan="2">Phase III: Implementation and Performance Evaluation</td><td>Phase III: Implementation</td><td rowspan="2">NA</td></tr>
<tr><td>Phase IV: Evaluation</td></tr>
</table>

[a] See Appendix 3-A for discussion of other watershed management frameworks

[b] Not applicable

case study could be used to fullest advantage, both in the watersheds that were studied and in other settings where similar methods could be applied.

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APPENDIX 3-A

DISCUSSION OF EXISTING FRAMEWORKS THAT HAVE BEEN APPLIED TO WATERSHED MANAGEMENT

Table 3-1 presents a typology of frameworks that have been applied to the processes of watershed assessment and management. This appendix discusses the frameworks listed in each of the four cells of the typology, and it presents several applicable flow diagrams that serve as background for the design of the conceptual approach presented in Figure 3-1.

Situational monitoring or assessment frameworks

Several frameworks pertain to monitoring or assessment that provide information for decision-makers but do not include the decision-making process. ERA, per U.S. EPA's *Guidelines*, is described in Section 2.1 and diagrammed in Figure 2-1. ERA is a situational process for decision support; it is initiated in response to past, ongoing or potential future adverse effects to ecological resources. ERA emphasizes the natural sciences and the separation of science and policy. Stakeholder involvement may be important for development of management goals during planning and, debatably, for problem formulation, but is considered inappropriate for analysis and risk characterization. The results of risk characterization are communicated to risk managers, but decision-making occurs outside the ERA process.[1] A *Framework for the Economic Assessment of Ecological Benefits* has been described by U.S. EPA[2] which explores the potential integration of ERA and economic valuation techniques; it has not been applied to watershed management but is included in the typology as a point of reference.

Environmental monitoring is an essential component of watershed management, and decisions about what to monitor implicitly are decisions about management. Most monitoring programs are limited to the collection of natural science data, but some include economic and institutional indicators as well. An example of the former is USEPA's Environmental Monitoring and Assessment Program (EMAP), which estimates status and trends of selected ecological resources by monitoring indicators of ecosystem structure and function and by measuring relationships between environmental stressors and impacts.[3] An example of a broader indicators framework is one developed by the Organization for Economic Cooperation and Development (OECD).[4] The DPSIR framework (see Table 3-1) calls for indicators of the social and economic conditions that drive environmental changes, and the policy and management responses to those changes, in addition to indicators of the environmental changes themselves. Monitoring system design usually stresses input from managers but not other stakeholders. For example, EMAP's indicator development process borrows several concepts (such as ecological values, assessment questions, and conceptual models) from the ERA *Framework* but does not assume stakeholder involvement.[5,6]

Regular monitoring or assessment frameworks

ERA generally is not a regular process; while its steps may be reiterated as more is learned, it is not intended to be continuous. Frameworks for the set-up of monitoring systems, including indicator design, usually depict a one-time (i.e., situational) process as well. However, a cyclical (i.e., regular) redesign process can allow monitoring systems to adapt as knowledge and management needs change.[7]

Situational planning and management frameworks

The Society for Environmental Toxicology and Chemistry has described an ecological risk management framework composed of the following steps:[8]

- issue identification
- goal setting
- management options development
- data compilation and analysis

- option selection
- decision implementation
- tracking and evaluation.

The process is informed by stakeholders during goal setting, and effective communication with stakeholders throughout the process is considered important. It assumes that economic analysis will be involved in the decision, but processes for integrating ecological and economic aspects are not discussed.

The *Framework for Environmental Health Risk Management* depicts a process that is similar, albeit with a slightly different ordering of steps (Figure 3-A-1).[9] Active engagement of stakeholders is encouraged throughout the process, and it is suggested that stakeholders be empowered to make decisions where allowable. While the framework is pictured as cyclical, it should be viewed as situational (responding to problems) yet amenable to adaptive management as necessary to implement effective solutions. A panel convened by USEPA's Science Advisory Board (SAB), tasked with making recommendations on the integration of environmental decision-making, presented similar ideas[10] but depicted the process more appropriately as unidirectional, albeit with feedback loops, rather than cyclical (Figure 3-A-2).

The U.S. Army Corps of Engineers (USACE) uses a six-step planning process for civil works projects, including those related to water resources and watersheds:

- Step 1 – Identifying problems and opportunities
- Step 2 – Inventorying and forecasting conditions
- Step 3 – Formulating alternative plans
- Step 4 – Evaluating alternative plans
- Step 5 – Comparing alternative plans
- Step 6 – Selecting a plan.

The process includes decision-making but in most instances does not include implementation, retrospective evaluation or adaptive management. Stakeholder involvement is intended to play

FIGURE 3-A-1

Framework for environmental health risk management (from PCCRARM[9])

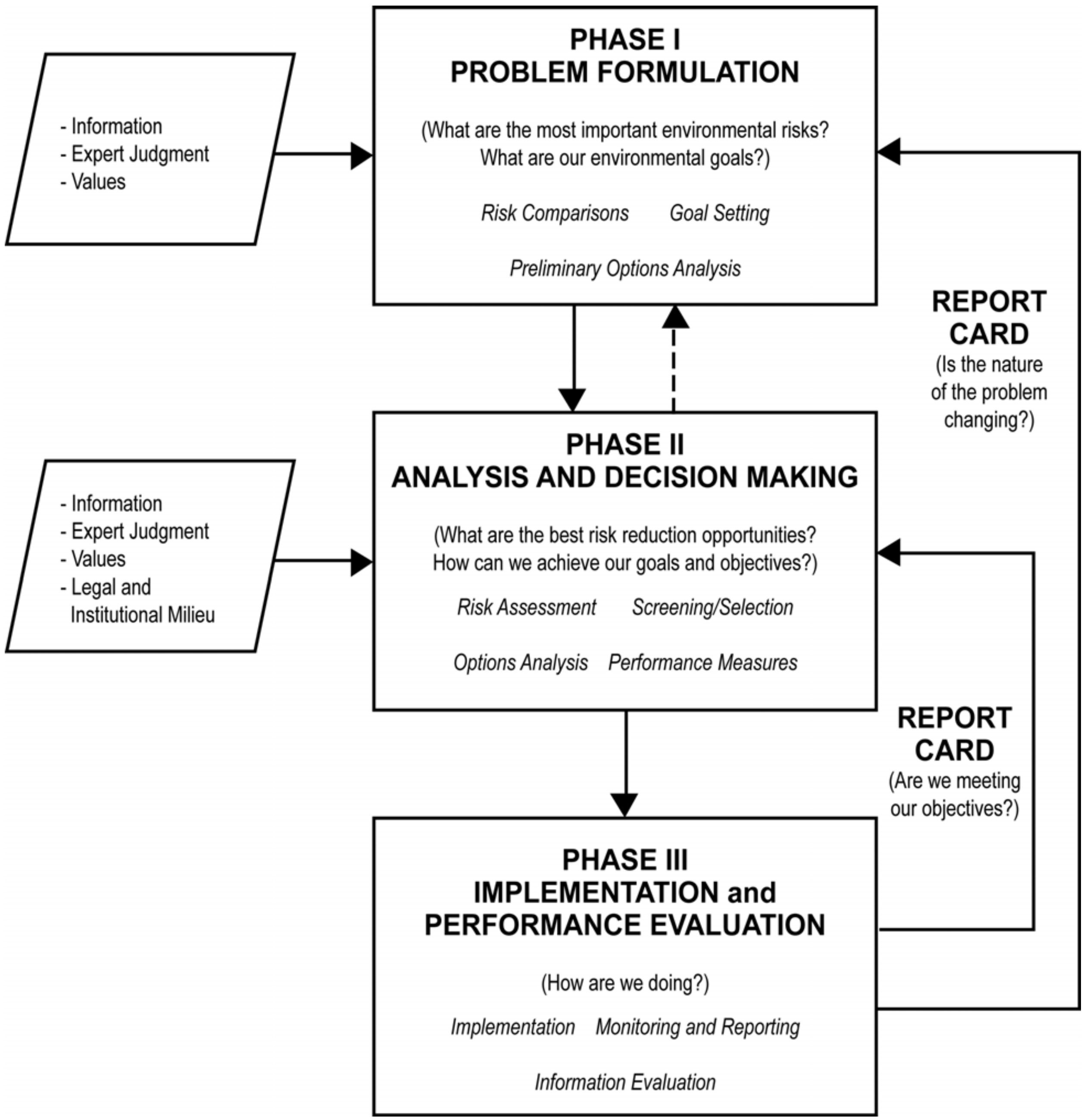

FIGURE 3-A-2

Framework for integrated environmental decision making (from SAB[10]) an important role in step 1, including in the selection of decision criteria; communication channels are to be maintained throughout the process; and stakeholder consultation is to occur after evaluation is completed and before plan selection. Evaluation of alternative plans includes quantifiable national and regional costs and benefits as well as nonquantifiable environmental and social impacts or benefits.[11]

By comparison, a planning and project development framework developed by the World Commission on Dams[12] provides for a more extensive stakeholder role and for adaptive management (Figure 3-A-3). Criteria ensuring, among other things, public participation, assessment of ecological risks, and consideration of a comprehensive set of alternatives, are checked at the conclusion of each development phase. Analyses of alternatives include the identification of people who are affected when lands or other resources are put at risk by the project, and negotiating rights with respect to the final decision are conferred according to risk burden. The framework emphasizes compliance with negotiated agreements during and post-construction. Finally, project operation is to be reviewed periodically and should adapt to changes in the project context.

A four-step process for the planning and implementation of watershed projects (Figure 3-A-4) was described by USEPA[13] and more fully elaborated by Davenport.[14] The process is designed to be carried out through a partnership of government agencies and local stakeholders, and it emphasizes involvement and action. The assessment and problem identification phase consists of four parts – inventory, analysis, problem identification and goal-setting – and is analogous to ERA. However, ERA assumes that analysis itself will require advance planning and substantial time and resources to conduct and will result in a quantitative characterization of risks, whereas the watershed project management approach emphasizes qualitative description of

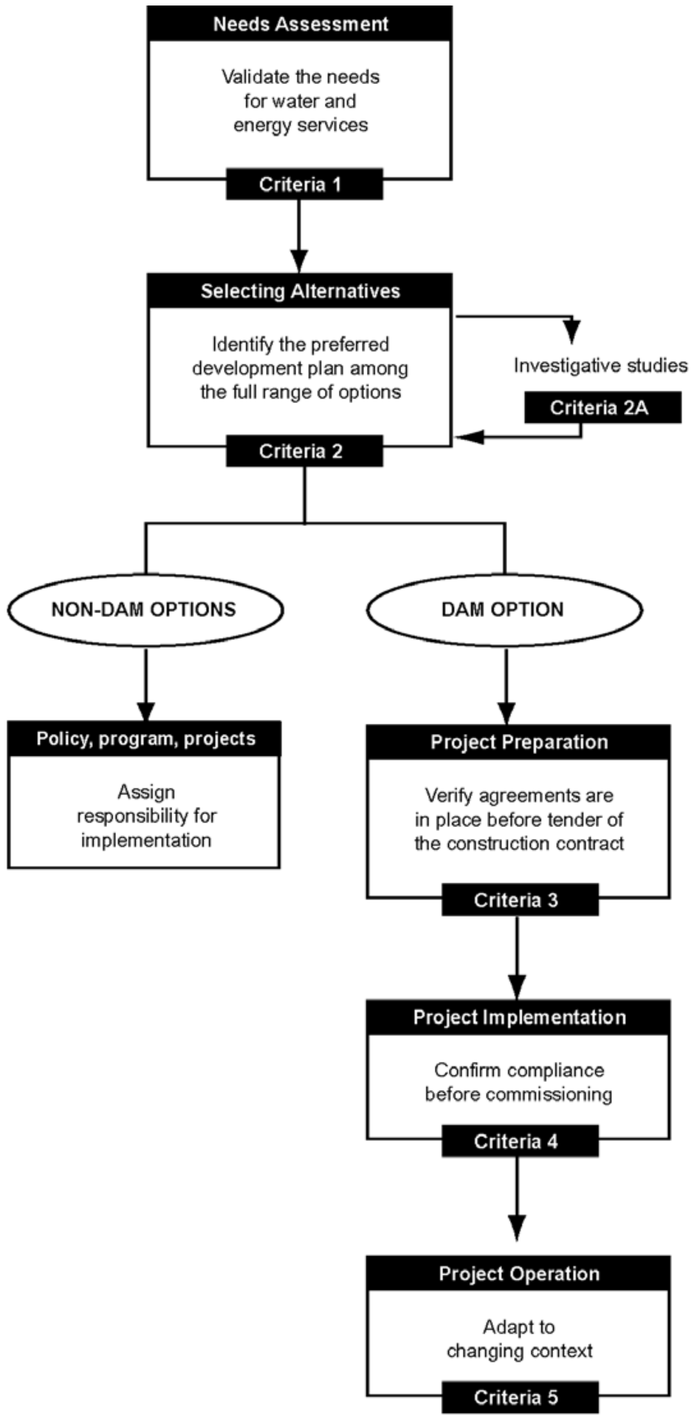

FIGURE 3-A-3

A framework for planning and project development of large dams, including five key decision points at which specific criteria should be evaluated
(redrawn from World Commission on Dams[12])

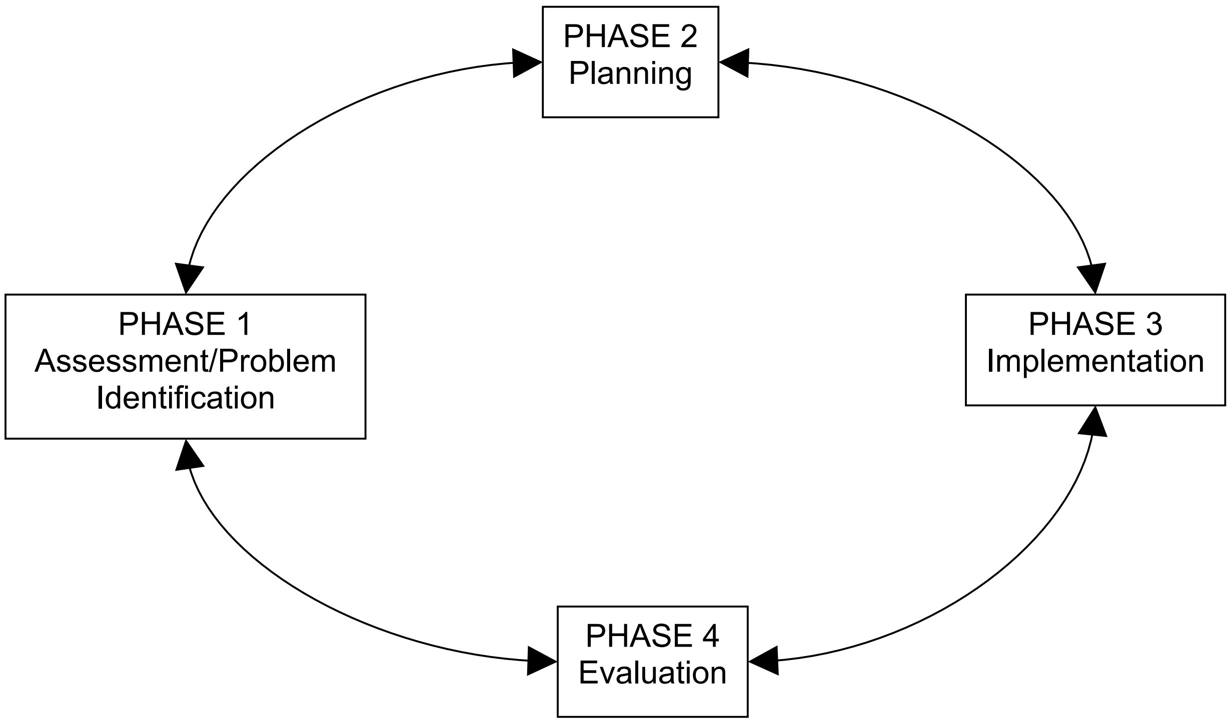

FIGURE 3-A-4

A watershed management model for the planning and implementation of watershed projects (redrawn from Davenport[14]) the most critical problems and their causes. Natural science is used to identify problems, and know-how, partnerships and consensus-building processes are used for making and implementing decisions. Project analysis, including economic analysis, is not emphasized. Like the *Framework for Environmental Health Risk Management* the process is pictured as circular; we have grouped it with situational methods on the assumption that efforts will conclude once conditions change. If a partnership is effective, however, an effort could be longstanding.

Regular planning and management frameworks

Several frameworks have been proposed for the regular and ongoing management of watershed resources. These regular processes can spawn situational analyses which may be portrayed as linked cycles.[15] For example, the U.S. Forest Service (USFS) uses a planning process (Figure 3-A-5) to guide the ongoing management of national forests and grasslands.[16,17] The spatial scale of planning ranges from national to regional to local, and it can be done at the watershed level if appropriate to the scope and scale of issues addressed. Existing plans authorize site-specific management actions, and outcomes are monitored and evaluated according to plan criteria in an adaptive cycle. New rounds of planning are undertaken after 15 years or as necessitated by issues or conditions. Stakeholders play an important role in the initial development of goals and are encouraged to participate in subsequent steps; participation opportunities are to be early, frequent, open and meaningful, and stakeholders may lodge objections before decisions are taken. Information development includes baseline analyses of both ecological and economic sustainability of current forest or grassland management practice. Ecological analyses include the effects of current or anticipated human disturbance (as compared to natural and historical human disturbance) upon ecosystem processes and system and species

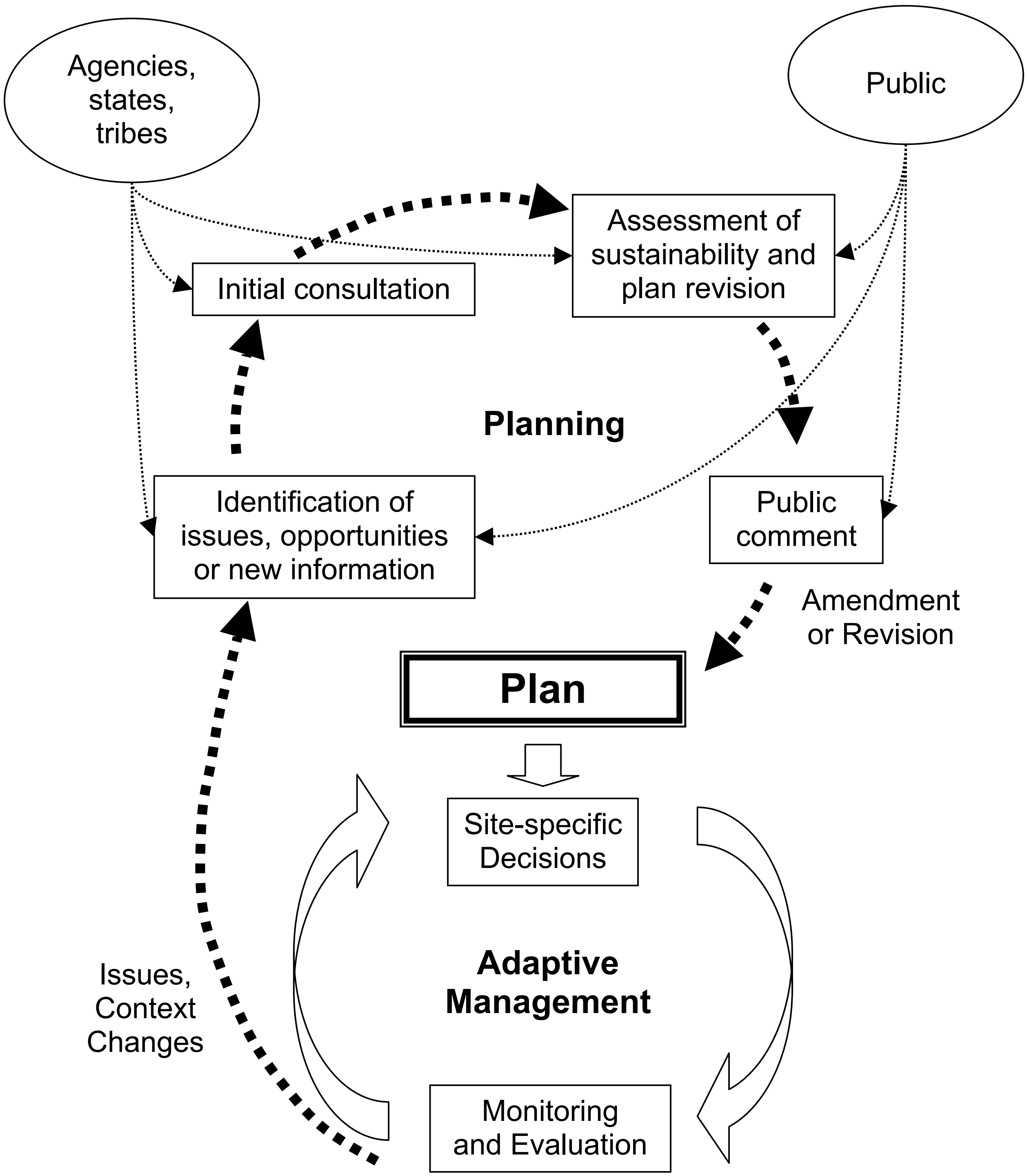

FIGURE 3-A-5

The USFS planning framework incorporates regular adaptive management and situational planning processes.

diversity. Social and economic analyses examine the benefits provided by forest lands, social and economic trends, and the society-forest relationship.

Many U.S. states have adopted a watershed management cycle, sometimes referred to as a "rotating basin approach," for implementation of the regulatory requirements and other programs of the Clean Water Act (CWA).[18,19] Whereas the approach usually is adopted to improve State agency efficiency, in most cases it has led to enhanced involvement of stakeholders as well, and the trend is toward more localized, partnership-based approaches driven by multi-stakeholder teams.[19] Typically, the state is divided into major watershed units, and CWA activities are implemented on a roughly five-year activity cycle that is staggered to begin in different years by watershed (Figure 3-A-6). The cycle begins with monitoring and assessment and continues through planning and implementation. "Assessment" as referred to here entails comparison of monitoring data and Water Quality Standards (WQS), a process which should detect likely adverse effects from stressors for which WQS have been determined but which falls short of risk assessment per se (see Section 2.3). While economic or other social-science studies are not precluded as part of this process, natural science is emphasized. In theory, activities such as review of designated uses, listing of impaired waters, issuance or review of point-source discharge permits, and award of loans and grants for water quality improvement projects are carried out in the implementation phase of this cycle, although in practice limited resources and competing priorities make this difficult to accomplish.[19] Total Maximum Daily Loads (TMDLs) may be developed and implemented for high-priority impaired waters; here the TMDL process is depicted as a situational cycle linked to the regular management cycle (Figure 3-A-6).

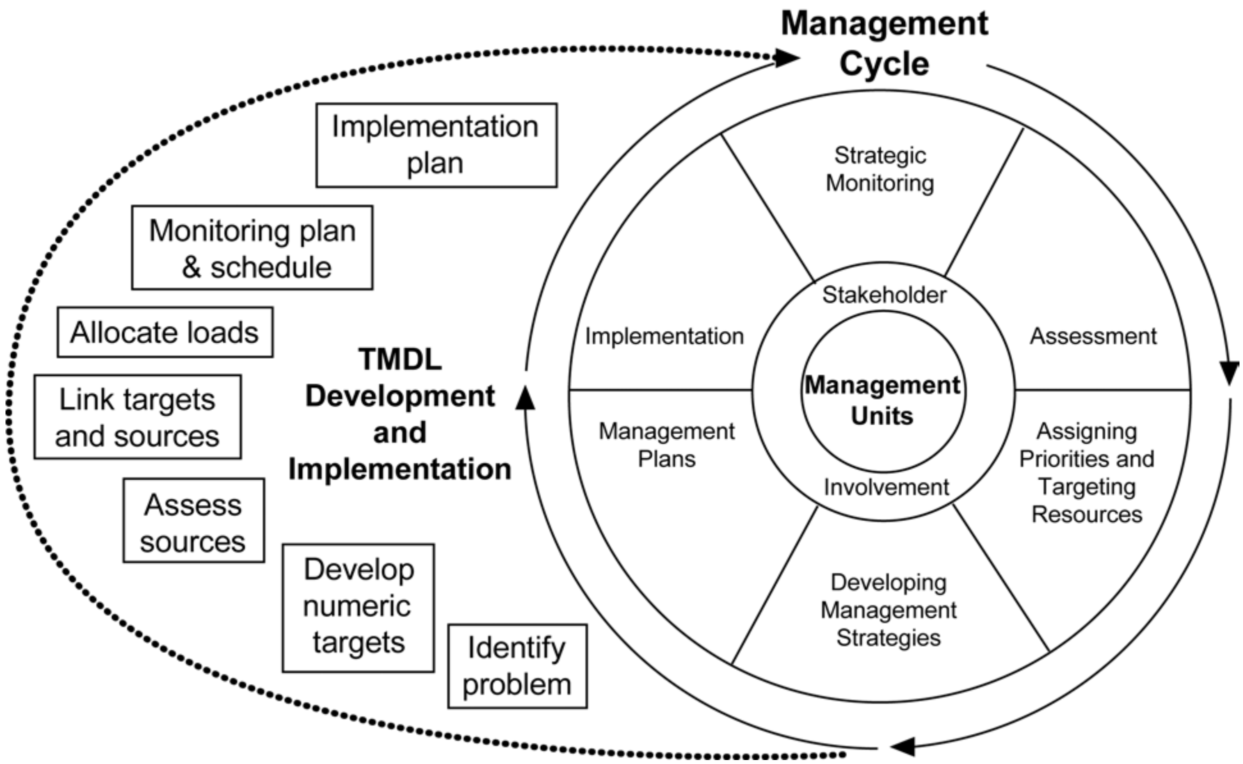

FIGURE 3-A-6

The watershed-based management cycle used by many states may include TMDL development and implementation (Adapted from USEPA[18])

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4. EVALUATING DEVELOPMENT ALTERNATIVES FOR A HIGH-QUALITY STREAM THREATENED BY URBANIZATION: BIG DARBY CREEK WATERSHED

A vision for integrating ecological risk assessment (ERA), economics and watershed decision processes has been presented in the previous chapter. The objective in this chapter is to consider a case study in which certain elements of that conceptual approach (see Figure 3-1) are implemented and field-tested with specific data. A large watershed in central Ohio, the Big Darby Creek, provides the locale and basis for the study design.

In 1993, the Big Darby Creek watershed was selected by the U.S. Environmental Protection Agency (USEPA) for one of five watershed ecological risk assessment (W-ERA) case studies for several reasons: the substantial interest by organizations at the local, state and federal level in protecting the watershed; the outstanding character of the aquatic biological resource; the range of sources and stressors (agricultural nonpoint sources, urban nonpoint sources, permitted discharges, etc.); the existence of a large, multiple year, watershed-wide database; and a commitment by Ohio EPA (OEPA) to co-lead the risk assessment team.

In 1999, while the W-ERA was in the later stages of completion, a USEPA-funded study was initiated by Miami University with the goal of integrating ERA and economic analysis to further inform environmental management efforts in the Big Darby Creek watershed. The methodological framework for this integrated research was rooted in a broadly based approach to sustainability that encompasses, but extends beyond, ERA. This approach views economic development as complementary with, rather than antagonistic to the maintenance of non-renewable resources. As such, it argues that sustainable systems require coordination between ecological, economic, and social considerations in order to maintain overall system resilience.[1]

Because the Miami University study was initiated well after the Big Darby Creek W-ERA, using information from the latter but carried out by a separate team, the two efforts were not integrated in an ideal sense (see Section 1.5.1). However, the research approach used illustrates some of the advantages, as well as the difficulties, of integrated study. Section 4.1 describes the watershed setting, and Section 4.2 discusses the W-ERA effort and its findings. The Miami University study is presented in Section 4.3, and Section 4.4 discusses these findings in light of the larger integration problem.

4.1 WATERSHED DESCRIPTION

Big Darby Creek is a high-quality, warm water stream located in the Eastern Corn Belt Plains ecoregion of the Midwest (Figure 4-1). The watershed encompasses 1443 km^2 (557 mi^2) and is home to a diverse community of aquatic organisms including many rare and endangered fish and freshwater mussel species. The Big Darby Creek watershed was given a conservation priority by The Nature Conservancy (TNC) through its recognition as one of the "Last Great Places" in the western hemisphere.[2,3] The risks to ecological resources in the Big Darby watershed derive from ongoing changes in agriculture and suburban land use.

The watershed drains portions of six counties in rural Ohio just west of Columbus. Agriculture currently comprises 92.4% of the land use of the watershed. Cropland, most of which is actively row-cropped, is the highest use (72%), followed by livestock pasture (8.6%). However, suburban Columbus is expanding westward in the Big Darby watershed. Currently, the western tributaries drain agricultural lands almost exclusively, whereas the eastern

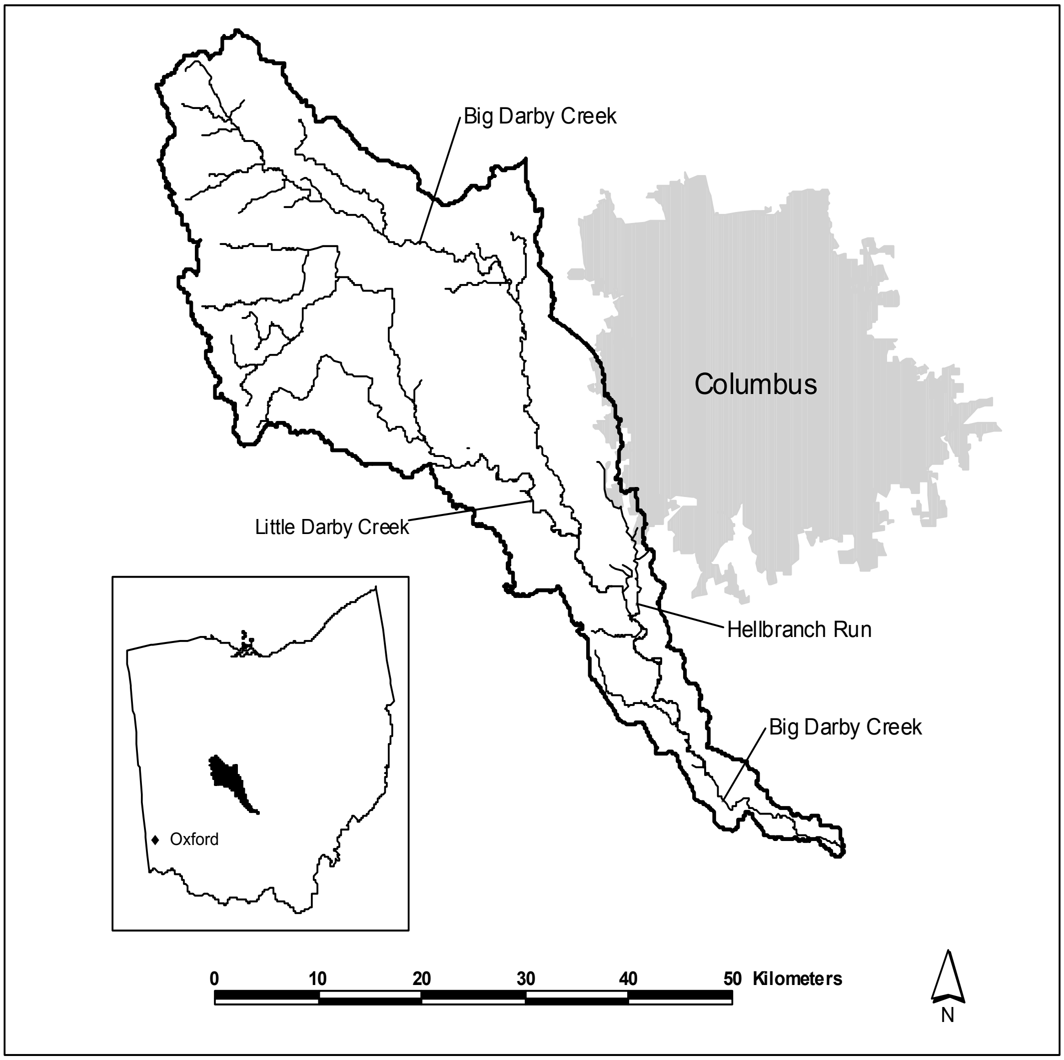

FIGURE 4-1

The Big Darby Creek watershed in central Ohio, USA. The Columbus metropolitan area is expanding into the easternmost area of the watershed, where Hellbranch Run is especially affected. Respondents surveyed in this study were drawn either from the watershed area, Columbus, or Oxford.

tributaries drain areas with increasing suburban and commercial/industrial land use. Urban development recently has quadrupled in some areas, with significant negative consequences for stream habitat. Although there have been recent improvements in fish and invertebrate indices in the Big Darby Creek mainstem, the easterly Hellbranch Run shows degradation.[3,4] A number of stream reaches in the watershed have been listed as impaired and are subject to potential regulation through development of total maximum daily loads (TMDLs, see Sections 2.1.2 and 2.3.1), mostly focused on phosphorous, nitrogen and sediment.

To the west from Hellbranch Run, the urban and industrial impacts are generally not greater than agricultural impacts, but given the present population of the region and the rapid rate of development, urban water pollution problems are a risk for a large part of the Big Darby watershed in the future. Without management, the increased frequency of damaging storm runoff and associated pollutant loads pose risks to the uncommon species, game fish and general aquatic system functioning. These are risks that could be reduced through best management practices for both urban and agricultural runoff.[5]

4.2 ECOLOGICAL RISK ASSESSMENT

The phases of ERA as described in USEPA's *Guidelines*,[6] i.e., planning, problem formulation, analysis, and risk characterization, are summarized in Section 2.1.1. This section describes the work that was conducted in each phase of the W-ERA for Big Darby Creek.

4.2.1 Planning

The OEPA database available for this assessment included standard water quality parameters such as suspended and dissolved solids, pH, oxygen-demanding substances, nutrients, ammonia and metals. It also included biological assemblage data describing the presence and abundance of fish species and of macroscopic sediment-dwelling invertebrates (termed benthic macroinvertebrates or benthos) collected by standard sampling procedures. Also available were a set of descriptors of stream corridor condition, including condition of substrates, instream habitat types (pools, riffles), channel stability and riparian zone vegetation. Multimetric indices that provided a composite assessment of habitat or biological quality, based on these data, included the Qualitative Habitat Evaluation Index (QHEI) for stream corridor condition; the Index of Biotic Integrity (IBI) and the Modified Index of Well-being (MIwb), which are measures of the functional and structural organization of the fish community, respectively; and the Invertebrate Community Index (ICI), which evaluates the structural organization of the macroinvertebrate community. These indices have been used extensively by the OEPA to establish biological criteria and to evaluate stream use attainment (see Section 2.3.1 and Appendix 2-B for further description of these indices).[7]

Cooperators in the Big Darby Creek Watershed Ecological Risk Assessment included the W-ERA team co-chairs from USEPA and OEPA and at various times representatives from The Ohio State University, The Nature Conservancy, the United States Geological Survey (USGS) and Operation Future, a conservation oriented farm group. Management goals for the risk assessment were developed through review of pertinent regulations, discussions with residents and resource managers, and meetings with the Darby Partners, a loose-knit group of over 40 public agencies and private organizations united by the shared goal of watershed protection. The overarching risk reduction goal from these discussions was to "protect and maintain native stream communities of the Big Darby ecosystem." Three specific objectives were seen as necessary to meet this risk reduction target:

1. Attaining criteria for designated uses throughout the watershed (see Section 2.3.1)
2. Maintaining OEPA's exceptional warm water criteria for all stream segments having that designation between 1990 and 1995
3. Ensuring the continued existence of all native species in the watershed.

The risk management problem was to ensure that these specific objectives could be met. The risk characterization would require understanding how various environmental factors might prevent meeting these objectives.

4.2.2 Problem formulation

Ecological assessment endpoints are measurable attributes of valued ecological characteristics. Two assessment endpoints were chosen for the Big Darby risk assessment:

1. Species composition, diversity and functional organization of the fish and macroinvertebrate communities
2. Sustainability of native fish and mussel species.

From a practical standpoint, the first of these endpoints could be evaluated utilizing three composite indices (IBI, MIwb and ICI) and the individual measures they comprise. It was determined ultimately, however, that while some of the available data were relevant to the second endpoint, the necessary information on life history and genetic diversity of native species in the watershed was not sufficient for evaluating their sustainability.[3] Therefore, only the first endpoint was carried further.

A critical step in problem formulation is the development of a conceptual model. It articulates the risk assessors' hypotheses on the relationships among the sources of stress, stressors, effects, and endpoints. Six significant stressors were identified for this watershed as affecting the assessment endpoint: altered stream morphology, increased flow extremes, sediment, nutrients, temperature and toxicants. A conceptual model illustrating the hypothesized relationship between land use, sources of stress, the aforementioned stressors, subsequent ecological interactions and the stressor signatures (i.e., characteristic changes in aquatic community metrics) is presented elsewhere.[3]

Seven risk hypotheses were developed based on the relationships inherent in the conceptual model:

1. No differences exist in community structure and function among the subwatersheds
2. No differences exist in community structure and function among time periods
3. Community structure and function will decline downstream from identified point sources
4. An increase in certain land uses or land use activities will result in a change in the IBI and/or the ICI
5. An increase in certain land uses or land use activities will result in an increase in the intensity or spatial or temporal extent of in-stream stressors
6. An increase in the intensity, or spatial or temporal extent of in-stream stressors will result in a change in the biological community as quantified by ICI and IBI metrics and species abundances
7. The pattern of response of the stream community can discriminate among the different type of stressors.

The first two were null hypotheses; analysis would determine whether they could be statistically rejected. The other five were maintained hypotheses, thought to be true; analysis would seek confirmatory or contradictory evidence.

4.2.3 Current status of analysis and risk characterization

The analysis to test these hypotheses was carried out in two phases. Hypotheses 1 and 2 were tested by analyzing historical biological assemblage data within the Big Darby Creek watershed.[4] Both hypotheses were rejected because certain spatiotemporal differences were shown in the analysis. Time series analysis, which was feasible for fish community metrics and IBI within the Big Darby Creek mainstem, indicated a general improvement over the time period 1979 – 1993. At the same time, spatial comparisons among the Big Darby, Little Darby and Hellbranch Run subwatersheds revealed significant spatial differences for IBI, ICI and several component metrics. In general, the Big Darby Creek mainstem showed superior biotic condition; however, some of this difference could be attributed simply to its comparatively larger drainage area. After correction for drainage area, many differences disappeared, but the biotic condition of the urbanized Hellbranch Run remained lower than the mainstem according to several measures.[4] These findings, while encouraging for the watershed as a whole, were consistent with concerns that suburban encroachment threatens watershed ecological resources in the eastern portion of the watershed. However, without an ability to correlate biological condition with stressors of concern or their sources, these results were of limited value for assessing risks associated with likely future changes in the watershed.

By contrast, hypotheses 3 to 7 required the analysis of point sources, land uses and stressors in spatial relation to biological data. Relatively few point sources of pollution are present in the watershed but most have shown negative effects on the mussel community for some distance downstream. Migration of species within the fish community making up IBI tended to remove the downstream effect. Thus, hypothesis 3 was confirmed for metrics focused on invertebrate species, but not for free swimming migratory species such as fish.

Initial attempts to analyze stressor effects derived from land use patterns were complicated by the watershed's relatively good water quality and higher than average IBI. The narrow range of variability in the biotic metrics and the chemical and physical parameters seen in the Big Darby needed to be assessed in the context of the greater variability of the region as a whole. Therefore, Norton et al.[8] analyzed biological, chemical and habitat data for the Big Darby Creek and other comparably-sized watersheds within the Eastern Corn Belt Plains ecoregion in Ohio, among which a wider gradient of the stressors and subsequent responses could be observed. Discriminant functions constructed using biological variables from this larger dataset were used to separate site groups into high-, medium-, and low-stress categories along stressor gradients. Analysis of the biological variables here did distinguish between higher- and lower-quality sites classified on the basis of six different types of stressors: degraded stream corridor structure; degree of siltation; total suspended solids, iron, and biochemical oxygen demand (BOD); chemical oxygen demand (COD) and BOD; lead and zinc; and nitrogen and phosphorus. Functions based on biological variables could also discriminate between sites having different dominant stressors.[8]

Using somewhat different methods for their data aggregation and analysis, Gordon and Majumder[9] analyzed similar data, but they also included land use (dense urban, forested or agricultural, as a percentage of each watershed) in an effort to develop regression models that could predict the ecological effects of future land use changes. A number of models showed some ability to explain average watershed IBI. For a set of 137 watersheds, the regression model explained 39.5 % of variance in the IBI when only stream corridor characteristics, land use and stream order were included (N = 467), 47.4 % when an index of chemical pollution stress was added to the model (N = 196), and 65.5 % when upstream IBI was added (to correct for spatial autocorrelation, N = 177). Percent dense urban land use was a strongly negative predictor. For the three models described, standardized regression coefficients for percent dense urban land use (which relate the variance in that factor to the variance in IBI) were -0.305, -0.258 and -0.179, respectively.

Therefore, hypotheses 4-7 were shown to hold true for the Eastern Corn Belt Plains ecoregion, and the relationships found can reasonably be applied in the Big Darby Creek watershed. These preliminary results suggested that fish and macroinvertebrate community responses to land use, stream corridor habitat and various chemical stressors are predictable to a degree. USEPA's efforts to apply these findings to the assessment of ecological risks in the Big Darby Creek watershed are still ongoing. Additionally, because of the identified impairments to some of its subwatersheds, Big Darby Creek is subject to the development of a TMDL by OEPA. Similarly, in an effort to assist planners, environmental organizations, government agencies, and concerned citizens, scientists and planners in The Ohio State University's City and Regional Planning Program, working on a USEPA-funded grant, have created an interactive, geographic information systems (GIS)-based screening tool to evaluate the biological effects of various changes within the Big Darby Creek watershed and other watersheds within the Eastern Cornbelt Plains ecoregion.[10]

4.3 ECONOMIC ANALYSIS

The overarching goal for Miami University's integrated ecological and economic analysis was to utilize the findings of ERA in an economic analysis that would be relevant to environmental management decisions in the watershed. At the time of initiating our integrated study in the Big Darby Creek watershed, the problem-formulation phase and early portion of the analysis phase of ERA had provided a clear picture of current conditions and apparent threats. Because the spatial scope of the analysis had to be expanded to all Eastern Corn Belt Plains watersheds in Ohio, a full complement of stressor-response or source-response relationships was not yet available, and the risk characterization had not been carried out. However, the following sections show that sufficient information was available for meaningful analysis.

The objective for this integrated case study was to undertake an analysis capable of informing decisions about reducing risks from suburban development. An independent modeling study sponsored by Miami University's Center for Sustainable Systems Studies[11] had quantified the range of effects on hydrology, sediment transport, and nitrogen concentrations from changes in land use. This study had considered three types of residential development in the Big Darby basin and had found that two different types of low density development protected the stream amenities very well. The analyses by Norton et al.[8] also informed the selection of stressors considered to be key influences on stream conditions following urbanization. Thus, the goal for this case study was an integrated evaluation of ecological and socioeconomic impacts associated with several land use approaches at the peri-urban fringe.

The specific objectives of the case study, therefore, were as follows: (a) to estimate the quantitative or qualitative impacts of a set of land use scenarios on stream ecological condition, local economic well-being and local quality of life, (b) to communicate these impacts to the public effectively, and to measure the overall economic value (see Section 2.2.2) corresponding to each scenario based on individual willingness to pay (WTP), and finally (c) to better understand the particular contribution stream ecological condition makes to the value of a given scenario.

4.3.1 Research approach

Based on prior work in the Big Darby watershed,[11,12] four development scenarios were used to compare outcomes for stream amenities: (1) a most-likely case of high density, conventional subdivisions using ¼- to 1-acre lots with water and sewer services discharging to the Big Darby; (2) a low-density ranchette development on 3- to 5-acre lots with local water and septic system disposal; (3) a low density cluster development, with intervals between clusters to achieve the same housing density as ranchettes (e.g., as maintained through purchase or set-aside of transferable development rights); and (4) a reference case of continued agriculture, which was the predominant land use pattern actually observed in the 1990s.

A dichotomous-choice contingent valuation method (CVM) survey instrument (see Section 2.2.2 and Appendix 2-A for a discussion of this method) was developed that allowed presentation of technical information on how changes in stream amenities are induced or avoided during land development, followed by expression of WTP for a certain outcome. Analysis was also carried out to develop a quantitative relationship between the four land use scenarios and stream biological integrity based on empirical relationships.

The survey approach involved in-person, multimedia presentations to noninteracting groups of 30-50 respondents who completed a questionnaire. The instrument was designed according to Arrow et al.[13] and implemented according to Dillman.[14] In addition, multiple stakeholders were brought into a pretest phase to gain insight about their viewpoints, as well as their suggestions about refining the survey instrument.

The survey/presentation was divided into three parts. The first section asked respondents their knowledge about and use of the Darby Creek before we provided information about this watershed. An example of such a question is: *"Do you believe some types of residential development lead to increased soil erosion and runoff of fertilizers and pesticides?"* In this case, 90% of respondents answered that they were aware of this issue. A follow-up question for those who answered *yes* to the above question asked: *"If so, do you think these runoff products do significant damage to streams and water quality?"* Again, a significant portion, 85%, of the respondents answered *yes*. Finally, those respondents who answered yes to both questions were asked: "*If so, do you think these runoff products will do damage to fish and other species in the stream*?" About 85% responded that they were aware of the damage of the runoff.

The second section was designed to engage the respondents as they were presented the effects that development might have on the environmental, social, and economic characteristics of the area. These effects are discussed below, but the reader must note that as this material was presented, respondents were asked questions about it. Many questions related back to the first section. For example they were asked: "*Did you know that the food base for many fish was tiny insect larvae that live on the stream bottom?"* About 30% of the respondents were not aware of this before the presentation. In another example, respondents were asked: "*Did you know that lawn and garden chemicals could affect the fish in the stream?*" Consistent with the results in the first section, 94% indicated that *yes* they were aware of this. The other 6% were made aware by the presentation. Thus it was possible to create a uniform minimum knowledge base across respondents. The final section was the valuation and demographics questions. These results are discussed in greater detail below.

The respondents were drawn from three different populations. “Residents” were defined as people who live within the study area of the Big Darby Creek watershed, both farmer and non-farmer. “Near-Residents” were people living outside the watershed but within the greater Columbus, Ohio metropolitan area. “Non-Residents” were drawn from people in the area surrounding Oxford, Ohio, a two-hour drive from Big Darby Creek. Residents and near-residents capture the value attached by people who use the area for residence or recreation (use value). These two groups also capture nonuse value if they value the watershed solely for the benefit of acknowledging its existence or are willing to contribute to preservation of future use. Non-residents may use the area for recreation, but at a much lesser rate than those in the Columbus area. The primary value for this group was expected to be nonuse value (see Section 2.2.2).

Samples were drawn at random from zip codes contained within each of the targeted areas. Respondents received a payment of $30 to cover their out-of-pocket travel costs and to show appreciation for the time spent in the one-hour presentation and survey response. Each respondent later received a mailed summary of survey results. The following sections develop the sequence of topics covered in the presentation to survey respondents and provide details on the valuation question.

4.3.2 Communicating the effects of urban development on ecological endpoints

Scientific understanding of the mechanisms by which residential development brings about change in streams generally can be reduced to four causal factor groups:

1. increased nutrients (which increase algal growth and affect the kinds of fish)
2. increased sediment (which decreases light penetration and affects the food chain)
3. increased toxic substances (which cause mortality in the food chain and fish)
4. increased runoff and flooding (which allow bank erosion and sedimentation).

The most difficult challenge for this project concerned the need to have the public (represented as survey respondents) understand the mechanisms inducing change in the stream well enough for them to attach value to the outcomes they prefer. This question of informing respondents about linkage mechanisms was addressed by presenting the following synthesis on watershed processes and ERA.

4.3.2.1 Increased nutrients, leading to change in fish species

Nutrients were described as chemicals that enhance growth of plants, on land as well as in the water. In high-density subdivisions, nutrients come from lawn fertilizers, from storm water runoff and, at some “downstream” locations, from household sewage. Runoff also carries soil and fertilizers from farmland, further enriching streams. The amount of nutrients entering the stream has been shown to depend on the number of people living in an area and how they manage their fields, lawns or gardens. Nutrient loading also has been shown to depend on the amount of hard surfaces (roads and roofs) developed in a neighborhood.

The main effects of increased nutrients on the amenities to be valued are nuisance-level growths of algae in streams. This increased growth can cause a change in water quality and the kinds of fish that live there. In enriched streams, fish species that feed on decaying stream-bottoms (many minnows, carp and catfish) are favored over those predator fish (e.g., bass and sunfishes) that feed on small fish. If nutrient input is very high, "fish kills" can occur. It was then possible to ask the respondents what types of fish they would rather have in a stream, and they were given the choice of (1) minnows, sunfishes and carp, or (2) bass, sunfishes and darters.

4.3.2.2 Increased sediment, leading to a decreased insect food base in streams

Survey respondents also were told and shown that the amount of sediment entering a stream from residential areas or farmland can vary widely, but poses a serious problem. During initial construction, erosion from bare soil can be very high during heavy rainfall. Large amounts of soil can enter the stream and remain there for years, despite being mobilized after every major rain event. After construction, less sediment enters from residential areas than from agricultural land that exhibits standard row crop cultivation, seasonally bare soils, and livestock wading in and alongside streams.

The main effect of sediment is to decrease the quality of fish and invertebrate habitat by filling small spaces between pebbles in the stream bottom that are normally home for insect larvae.[15] Such insects are the main food for many types of fish, part of the rich ecological diversity in the Big Darby Creek. Without these insects the number and kinds of fish decrease. Furthermore, very fine particles are shown to stay suspended in the flowing water, making it cloudy and also decreasing the ability of fish to find their prey.

4.3.2.3 Toxic substances, changing the insect food base, fish species and causing disease

In the section on effects from toxic substances in runoff, the survey respondents were given information on how storm runoff washes pesticides from cropland, lawns or gardens into streams. Such compounds also may come from spilled oil, gasoline, and other automotive chemicals present on roads and driveways. These chemicals often cause change in the numbers and kinds of fish in streams, favoring fish that tolerate these substances. Respondents were asked whether they knew that lawn and garden chemicals could affect the fish in streams.

4.3.2.4 Changes in runoff and flooding patterns, decreasing habitat quality and causing a shift to fewer, more tolerant species

Natural streams were described as having bends, pools and riffles, with logs and limb "dams" all the way to their headwaters, thus slowing passage of water. Slow natural drainage from the land also allows water to seep into the ground slowly after heavy rains, replenishing ground water. However, with some residential development, streams are straightened, logjams are removed and storm water drains quickly off the land, increasing the risk of downstream flooding.

The effects of these physical changes were described in the presentation as increasing the speed of water flow, causing further erosion from stream banks and increasing flood heights. After the runoff, water flow can become quite low in the absence of a strong groundwater recharge. These alternating high and low flows drastically change the quality of fish habitat, reducing biological diversity. Instead of many different depths and bottom types, the channel becomes wide and shallow. During low-flow periods, water moving over or through a gravel base becomes too shallow to be inhabitable. The resulting crowded conditions lead to increased death rates for fish as they use up nearly all of the available oxygen.

4.3.3 Communicating the effects of urban development on economic and social services

While the information above sought to frame certain values attached to the Big Darby, respondents also derive other kinds of value from economic and social functions within the area.

To isolate the value placed on ecological services, one must control for value related to the economic and social services. Accordingly, the economic and social dimensions included in a sustainable development framework[1] also were briefly described.

In considering the value of economic services, the dominant endpoint is increased economic well-being. Although many measures that contribute to economic well-being were considered, the presentation focused on four economic outcomes: (1) dependence upon agricultural employment; (2) distance to employment for non-agricultural workers; (3) provision of retail services; and (4) impact on the local income base. Employment opportunities for agricultural and non-agricultural workers can be expected to change significantly across the four development scenarios. As residential development increases, agricultural employment opportunities will decline, but there would be sufficient population growth to justify expansion of retail services. Dependence upon commuting for non-agricultural work not only involves travel costs and value of time dimensions, but also has feedback effects as commuters either make purchases outside of the Big Darby or, conversely, bring higher incomes back to the local area. This is one of the ways in which development would be expected to affect the local income base. In addition, the income profile of residents who would be expected to populate the study area would vary under the different scenarios. Questions were included to capture respondent preferences about these economic outcomes.

With respect also to social services, the ultimate endpoint is increased quality of life. Among the many factors that contribute to quality of life, the presentation focused on four social outcomes: (1) open space, (2) privacy, (3) public services, and (4) quality of education. These factors vary among the different development scenarios as well. The change in open space and privacy during the transformation from rural to suburban could be a confounding variable of importance to respondents. As residential development progresses, the availability of open space for use in recreational activities and the degree of privacy begin to decline. In addition, residential development not only brings a need for increased public services, such as police and fire services, but also difference in access as the proximity to these services changes. Moreover, the quality of publicly provided elementary and secondary education is likely to change with increases in local income and property wealth, and as voter tastes for education change. Questions were included to capture respondent preferences about these social outcomes.

4.3.4 Land use scenarios for framing expression of preference and value in the stream

All the variables considered in the previous two sections vary among the land management or development options, allowing an approach that estimates stakeholder value through CVM surveys. The CVM questionnaire tries to focus on the unique amenities that could be at risk while acknowledging that other factors come into play. The information provided to survey respondents about physical stressors and ecological, economic, and social mechanisms can affect the estimate of WTP in terms of the direction and magnitude of the potential bias.[16,17] Thus, the survey instrument must have questions concerning preferences as well as values. To facilitate an understanding of the contrasts in options and outcomes, maps, data and photographs were used to frame WTP to conserve amenities described in each of the development scenarios below.

For easy reference, survey respondents were provided Table 4-1a and Table 4-1b which show the levels of effect from each of the objective factors considered in the section on linking mechanisms. In each case, a range of possible effects was described, categorized as low,

TABLE 4-1a

Relative effect of four housing development scenarios on the four main causes of change in Big Darby Creek

	High Density Development	Low Density Ranchettes	Low Density Clusters	Agriculture
Nutrient input	Medium to high	Low to high	Low to medium	Medium to high
Sediment input	Low to high	Low to medium	Low to medium	Medium to high
Toxin input	Medium to high	Low to high	Low to medium	Medium to high
Change in flow patterns	High	Low to medium	Low to medium	Medium to high

TABLE 4-1b

Relative effect of four housing development scenarios on socioeconomic outcomes in Big Darby Creek

	High Density Development	Low Density Ranchettes	Low Density Clusters	Agriculture
Economic Outcomes				
Agricultural employment	Low to Medium	Low	Medium to High	High
Retail services	High	High	Medium	Low
Distance to employment for non-agricultural workers	Low	Medium	Medium	Medium to High
Local income base	High	Medium	Medium	Low to Medium
Social Outcomes				
Open space	Low	Medium to High	Medium to High	High
Privacy	Low	High	Medium	High
Proximity to police and fire services	High	Medium	Medium	Low
Quality of education	Medium to High	Medium	Medium	Low

medium, or high in both the script and color slides. These categories are intended to reflect increasing levels of risk. For instance, low nutrient input would be that input leading to nutrient concentrations in the stream that are in the range of the lowest 1/3 of the observed data on nutrient concentrations. The factors are normalized such that when the effect reaches high levels, there is risk to stream integrity.

4.3.4.1 High density development

The base case against which the respondents are asked to indicate preferences or WTP (to avoid) is illustrated in Figure 4-2a. It shows a 4-mi^2 area that includes both sides of the Big Darby, not far off I-70. It represents the conventional residential development that many people expect based on the patterns already being seen in the Columbus area. The characteristics defined into this high-density scenario are: 15% open or agriculture, 70% residential, 10% forest, and 5% nature preserve. The lot size is about ¼ to 1 acre and the residential density is 200 dwelling units per 100 acres of land.

Nutrient input is affected by storm water runoff that carries lawn fertilizers at certain times of the year. In this scenario, the aggregate effect is expected to be medium to high. Sediment input from this scenario will be high during construction periods, but then may be fairly low. Toxin input will be medium to high depending on lawn and garden care practices, and whether a storm water treatment system is in place to treat the chemicals scavenged from roads and driveways. The pattern of stream flow, flood frequency and scouring is changed considerably, mainly due to the very large increase in hard surfaces. The respondents were asked several questions as to their preferences for avoiding associated enrichment, toxins and extreme flow outcomes.

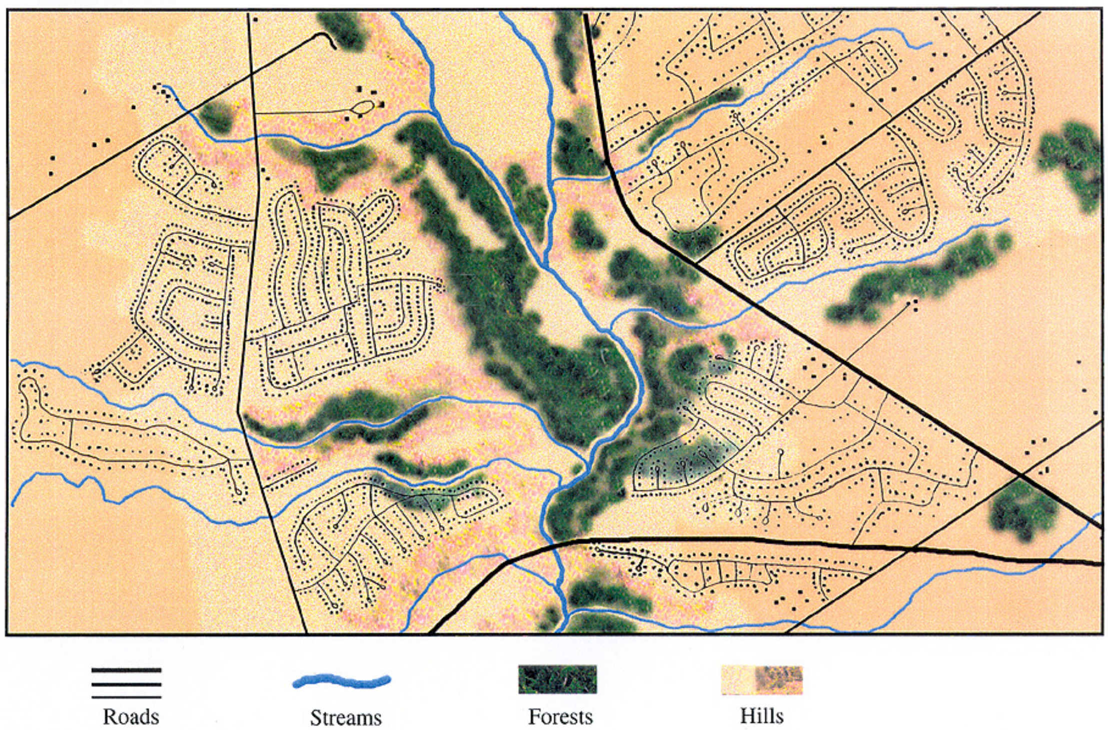

FIGURE 4-2a

Illustration of high density scenario
(dots represent houses)

4.3.4.2 Low density ranchette development

A second scenario, shown in Figure 4-2b, illustrates the same 4-mi^2 area but with development in the form of large lots, based on the patterns already observed in many suburban areas. The characteristics defined for this "ranchette" residential development were: 10% agriculture, 70% residential, 15% forest, and 5% natural preserves. The dwelling unit density is 20 units per 100 acres, with 3 to 5 acre lots.

The inputs of nutrients and toxins can vary from low to high in this scenario depending on how much of each lot is left in natural vegetation and how the lawns are maintained. Some nutrient input to the stream from septic tank seepage also is possible. When few pesticides are used on lawns and much of the land is left in a "natural" state, then both nutrient and toxin input will be much lower than in the high-density scenario. However, when large areas are maintained as lawns using standard chemical lawn treatments, then both nutrients and toxins could be almost as high as the high-density scenario.

Sediment input also will range widely, from low to medium, with some entering the stream mainly during the construction phase, and tending to be much less over time. Changes in stream flow peaks will be low to medium, and much less than the high-density scenario. In this scenario, stream habitat will depend largely on the amount of forest and wetlands left near the stream channel. In comparison to conventional agriculture, however, the overall change in the Big Darby system from large lot development is likely to be positive. The survey respondents were asked whether it is likely that residents of this ranchette type of development will leave enough land in its natural state to protect Big Darby water quality, and whether they would be

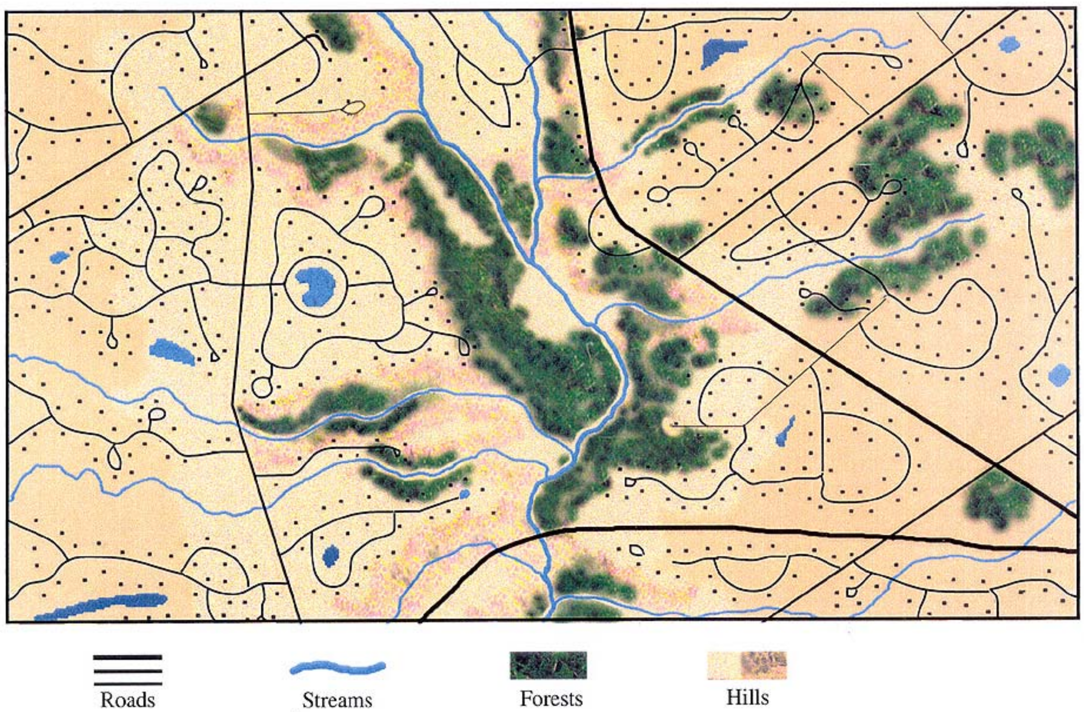

FIGURE 4-2b

Illustration of low density ranchette scenario
(dots represent houses)

willing to pay slightly higher land and construction costs to guarantee that sediment input to the creek is minimized by erosion barriers and sediment traps. They also were asked whether taking over nearly all the farmland is a significant negative consideration for them.

4.3.4.3 Low density cluster development

A third scenario, shown in Figure 4-2c, illustrates the same 4-mi^2 area, but with a clustered development that keeps most of the land in agriculture. The characteristics defined into this type of development are: 60% agriculture, 20% residential, 15% forest, and 5% nature preserves. The dwelling unit density is 20 units per 100 acres, the same as for the ranchette development.

Nutrient input from this scenario is shown to vary from low to medium depending primarily on associated farming practices. The cluster housing developments would each include their own sewage treatment system, possibly in the form of package treatment and wetland wastewater application, with little input to the creek. Maintenance of lawn area also would contribute little because of the small lot sizes for housing. Nutrient input from farms may be insignificant, depending on fertilizer applications and the density of livestock.

Sediment input will vary here much as it does in the ranchette scenarios, with higher input during construction, decreasing with time. Because the amount of bare land in hard-surface roadways is less than in either of the other two residential scenarios, overall sediment input even during construction will be low to medium, with the input determined by the amount of land left in agriculture. Soil-conserving agricultural practices such as low-tillage could decrease the sediment load even further. Toxin input will be lower in this scenario than for either of the other residential developments because of the smaller area of lawns and hard

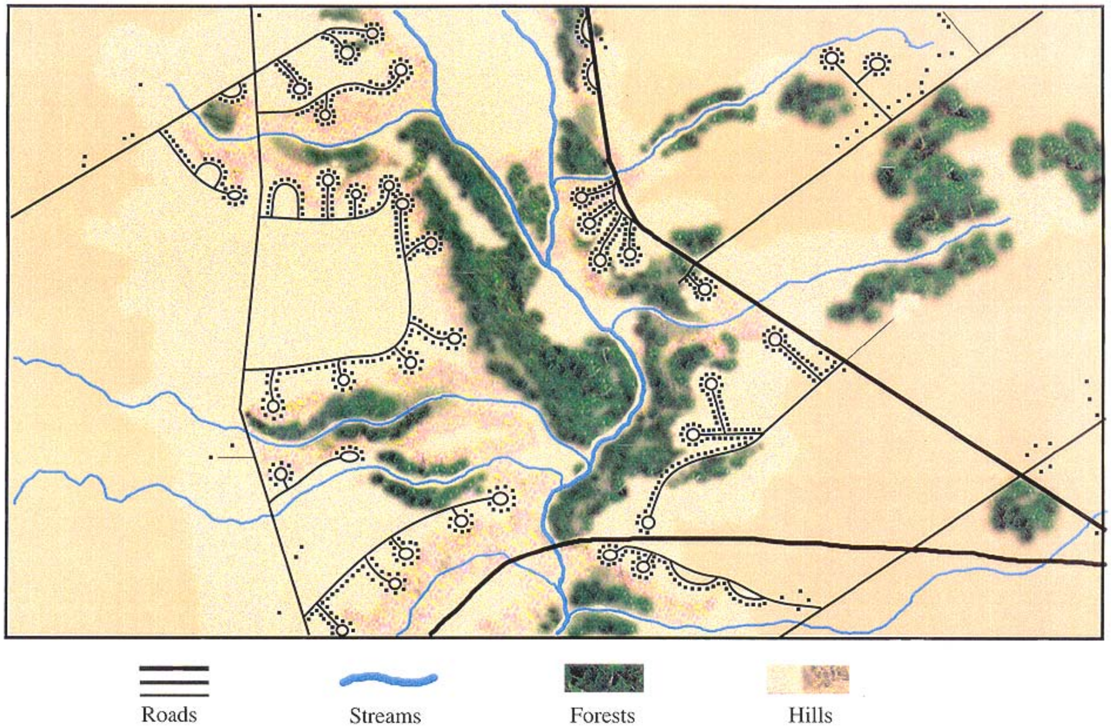

FIGURE 4-2c

Illustration of low density cluster scenario
(dots represent houses)

surfaces, but the range of agricultural practices largely will determine the level of toxins reaching the stream. The altering of stream flow and flooding pattern is lower here than for the agriculture or high density scenarios.

4.3.4.4 Agriculture land use

The final scenario is shown in Figure 4-2d. This scenario shows the land use and residence density actually observed in the area in the early 1990's. The characteristics of this "present landscape base case" are: 75% agriculture, 10% "residential" (including farm lawns), and 15% forest. The dwelling unit density is 2 units per 100 acres.

The input of nutrients, sediment, and toxins in this scenario can be medium to high, depending on local agricultural practices and the amount of livestock (see Table 4-1a). The time of cultivation and the amount of fertilizer and pesticide application also influence the amount of sediment, nutrients and toxins in runoff reaching the stream. Certain farming practices can be adopted to reduce fertilizer applications and minimize runoff after rain events. However, many farmers in the Big Darby drainage area already use conservation tillage practices to reduce nutrient, pesticide, and sediment inputs.

The altering of stream flow characteristics under this scenario is medium to high (relative to a pristine, unfarmed condition), also depending on farming practices. Because the Big Darby area is fairly flat, water does not flow to the stream quickly, and farmers are often anxious to drain the water off their fields. Tile drainage systems and straight clean waterways have been introduced locally, increasing water flow and transport of nutrients off the land. The survey respondents were asked how important it is to them that a large portion of the Big Darby watershed be retained in agricultural land use.

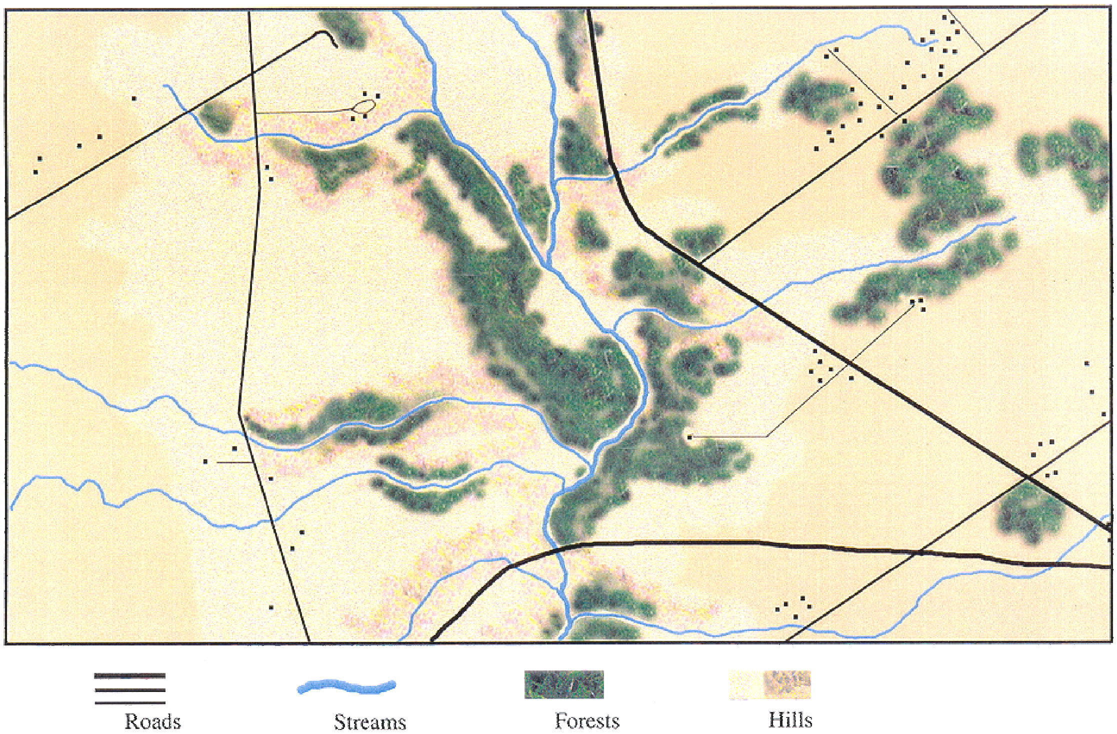

FIGURE 4-2d

Illustration of present agriculture scenario
(dots represent houses)

4.3.5 Eliciting monetary valuation

The four scenarios, and the ecological, economic, and social variables affected by residential development in the hypothetical 4-mi^2 area, were presented visually to groups of about 30 respondents, who completed the survey questionnaire during several pauses in the presentation. In the first part of each session, respondents were introduced to the potential impact of development under each of the scenarios. Photographs taken within the Darby watershed were used to illustrate these effects.

In the latter part of each session, respondents were asked to identify which of the four scenarios they felt were most likely to occur and which they most preferred. This was followed by a WTP question used in the CVM analysis. A map showing a portion of the Big Darby Creek watershed was displayed, with a 150-mi^2 area just west of Columbus highlighted as "facing likely development over the next 20 years." The Darby watershed sample was drawn from this area. Each respondent was then confronted with a choice between the high density base case and one of the other development scenarios. This question was framed around the idea that a group of citizens, along with government officials at both the local and state levels, had developed a fund to ensure that development in the highlighted area of the Darby follows a path that would lead to a specified state. It is proposed that monies for the fund would come from a hypothetical check-off on Ohio State Income Tax forms similar to current donation opportunities for wildlife and for natural areas. The respondents were asked if such a check-off were available, asking them to contribute $____ to the fund, would they check YES or NO? The dollar amounts were filled in by a random allocation within the questionnaire of amounts ranging from $1 to $100, based upon results from focus group pretests.

A method suggested by Loomis and colleagues[18] was used to calculate mean WTP based on survey results. For a particular landscape scenario, a core logit equation was formulated as follows:

$$\text{VOTE} = f\,(\text{FUND, INC, USEFREQ, AGE, Z}), \qquad \text{4-1}$$

where VOTE is a dummy variable indicating whether the respondent voted YES or NO on the WTP question (preferring an alternative to the high density outcome), FUND is the respondent's posed dollar value contribution, INC is household income, USEFREQ is the number of times per year the respondent or family uses the Big Darby for outdoor activities, AGE is the age of the respondent, and Z is a variable indicating special circumstances that might influence WTP. For example, one question asked whether the respondent or a family member considered themselves to be a farmer; another asked if the respondent was a member of an environmental group.[a] Alternate specifications of the model were estimated using different respondent variables as the basis for the core equation (i.e., Z was a dummy, YES/NO, variable either for "farmer" or for "environmental group member"), then separately considering status of the respondent (Resident, Near-Resident, Non-Resident), and finally by scenario type (Ranchette, Cluster, Agriculture). The results can then be interpreted as the contribution of each of the variables towards an individual's probability of contributing to the fund.[b]

Mean values for all the variables are used in conjunction with the estimated regression coefficients from the logit regression to estimate a mean WTP. The resulting general estimates from two alternate model specifications are shown in Table 4-2. The upper and lower bounds for

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[a] In the sample of 766 respondents, 83 stated they were members of an environmental group, 66 said they were farmers, 8 were both, and 625 were neither.
[b] Details of these results are available upon request (Loucks, Erekson, Elliott, McCollum, and Bruins, submitted to Landscape Ecology, 2002).

TABLE 4-2

Mean willingness to pay and confidence intervals for two model specifications[a]

Sample	Specification 1[b]			Specification 2[c]		
	Mean WTP	90% C.I. Min	90% C.I. Max	Mean WTP	90% C.I. Min	90% C.I. Max
Entire	$37.65	$28.64	$58.18	$37.96	$28.72	$58.94
Resident	$49.82	$29.29	$156.09	$51.44	$29.15	$162.39
Near-Resident	$33.91	$23.37	$68.28	$33.38	$23.40	$67.19
Non-Resident	$25.99	$14.99	$80.57	$25.45	$15.06	$75.11
Ranchette	$25.62	$17.15	$58.91	$25.19	$17.02	$54.47
Cluster	$67.05	$30.89	$261.17	$69.73	$27.33	$291.69
Agriculture	$29.58	$20.86	$57.72	$29.24	$20.50	$57.09

[a]Residents, n = 322; Near-Residents, n = 319; Non-Residents, n = 106
[b]Model specification includes dummy variable for "farmer"
[c]Model specification includes dummy variable for "environmental group member"

the 90% confidence intervals are estimated using a simulation model with 10,000 random draws of the estimated regression coefficients. As would be expected, the mean values are higher for residents than for near-residents, and those are higher than for non-residents. In addition, the WTP for a Cluster landscape alternative was significantly higher than for the Agriculture or Ranchette alternatives.

4.3.6 Linking stream integrity to the development scenarios

The approach to linking stream ecological condition with the development scenarios relied to some degree on an empirical relationship between impervious surface area (a runoff inducing condition) and IBI. Recent work by Yoder et al.[19] showed that for the lowest quartile of urbanization around Ohio stream sampling sites (with impervious surface of less than 4.3% of watershed area), modal IBI is 42. This is just above the Warm Water Habitat criterion of IBI = 40 (and well below the Exceptional Warm Water Habitat criterion of 50).[a] For the second quartile of urbanization (4.3 to 14.6% impervious), the modal IBI is 39.5. For the third quartile (14.7 to 29.3% impervious), the IBI is 35.0, while for the fourth quartile (over 29.3% impervious), the estimated mid-range IBI is 24, or highly degraded. This work suggests a likely median of 3 percent impervious surfaces for rural agricultural land, and 20 percent or more for urban areas, both reflecting a literal understanding of the term impervious surface: the total surface area of roads, driveways, and roofs. These results also suggest a possible threshold for serious degradation of IBI when impervious surfaces are at or above 20 percent. In addition, the

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[a] Under OEPA's designation, exceptional warm water habitat differs from warm water habitat in having an exceptional or unusual community of species when compared to reference sites (i.e., comparable to the 75th percentile of reference sites on a statewide basis). More stringent biological criteria are established for exceptional waters (see Section 2.3.1).

majority of watersheds having more than 15% impervious surface do not meet the OEPA's Warm Water Habitat Biocriteria.[19]

However, runoff hydrologists[20,21] have over many years developed an empirical relationship between modified surface conditions (such as cultivation, or residential lawn surfaces) and the intensity of runoff induced. These papers show that intensive cultivation creates runoff-inducing conditions in agricultural areas roughly equivalent to a moderate level of impervious surfaces. Using a transformation based on the "curve numbers" adopted by the hydrologists, a measure, "runoff-inducing condition," has been developed as shown in Table 4-3 that captures the conditions (and IBI) associated with each of the development scenarios.

TABLE 4-3

Runoff-inducing condition and IBI per scenario

Scenarios	Indicated Impervious Surface Assumptions (after Yoder et al.)	Interpolated Runoff-Inducing Conditions	Modal IBI[a]
Agriculture	3%	16.9	42
Ranchette	-	16.3	43.0
Cluster	-	17.0	41.8
High Density	20%	21.3	35

[a]Interpolated from a graph linking the results of Yoder et al.; interpolated runoff-inducing condition, and IBI. Details available from the authors.

4.3.7 Linking stream integrity and willingness to pay

There is a great deal of interest among environmental managers in determining the dollar values that may be associated with changes in ecological condition. When respondents expressed WTP to obtain one of the development scenarios, their valuation took into account the economic, quality-of-life and ecological ramifications of adopting that scenario in place of the expected, high-density scenario. In this case study, those ecological changes were quantified as units of IBI change. A multimetric index such as IBI has the potential to respond in complex fashion to changes in water or habitat quality. The large number of metrics it includes, however, and the functional complementarity among those metrics, apparently lend it a degree of numerical stability. In practice, IBI often has been treated as having cardinal properties for purposes of environmental analysis and regulation. In this section, the investigators probe the implications of their data for associating a dollar value with a unit of change in IBI.

Table 4-4 provides preliminary estimates of the relationship between WTP and IBI change in the 150-mi^2 area considered in the survey. For example, in the case of respondents considering the agriculture alternative, the change in runoff-inducing condition from high density to agriculture (from 21.3 to 16.9) corresponds to an IBI improvement of from 35 to 42. Respondents for the agriculture cohort had a mean WTP of $29.58, corresponding to the 7-point improvement in IBI. Thus, an estimate of the WTP per unit of IBI for this cohort would be $4.23 per unit of IBI. The corresponding estimates ($9.86 per unit of IBI) for the cluster cohort were more than double that of the agriculture cohort, and almost triple that for the ranchette cohort.

For many reasons, however, caution is necessary in interpreting these IBI-normalized WTP values, since these results do not separate changes in ecological and related risks from other environmental, economic and social changes associated with the development scenarios. In fact, since the IBI changes associated with these three scenarios were similar in magnitude, it is likely that the expressed differences in value between the scenarios were influenced by both non-ecological factors and certain perceptions about ecological factors not captured by IBI.

TABLE 4-4

Estimated WTP per unit of IBI improvement over a 150-mi^2 study area for two model specifications

	IBI Improvement	Specification 1		Specification 2	
		Mean WTP	Mean WTP/IBI	Mean WTP	Mean WTP/IBI
Ranchette	8	$25.62	$3.20	$25.19	$3.15
Cluster	6.8	$67.05	$9.86	$69.73	$10.25
Agriculture	7	$29.58	$4.23	$29.24	$4.18

Analyses now underway are looking more closely at the respective, marginal contributions of the ecological, economic, and social factors to WTP.

4.4 DISCUSSION

When the Big Darby Creek watershed ERA (Section 4.2) and economic analysis (Section 4.3) are considered collectively, the overall work has some of the ideal characteristics of an integrated analysis as described in Section 3.3 and diagrammed in Figure 3-1. It also demonstrates some of the problems that result when integration is not a goal from the outset.

Assessment planning involved a wide variety of partners and stakeholder groups, resulting in clearly defined goals and objectives (see Section 4.2.1). The problem formulation conducted as part of the ERA identified two ecological assessment endpoints, of which one could be feasibly measured, and conceptual models developed relating human activities in the watershed to stressors, to effects on endpoints, and to specific measures of effect. An analysis plan for the evaluation of specific risk hypotheses was developed and substantial progress was made toward the analysis and characterization of baseline risk. The ERA made use of data collected as part of the statewide watershed management cycle (see Figure 3-A-6) and had begun to provide empirical, stressor-response and source-response relationships that will be useful in TMDL development.

The team conducting the economic analysis formulated a set of management alternatives, in this case suburban development scenarios, focused on one of the more severe concerns identified in assessment planning and problem formulation: stream degradation linked to urban encroachment in the watershed's eastern portion. The subsequent steps, analysis and characterization of alternatives and comparison of alternatives, were similar in form to the example shown in Figure 3-3 but with a number of important differences. As shown in Figure 4-3, they provided a qualitative analysis of the effects of each scenario on a set of important stressors affecting instream biota and on economic and social services to watershed residents. They did not examine the financial costs or other market-based effects of the management alternatives. In that those costs would accrue to land holders who would have to forego valuable development options, the analysis also did not address equity.

To compare the alternatives, the ecological, economic and social impacts of each scenario were incorporated into an integrated CVM instrument. The comparison was effected using monetary WTP associated with each scenario. That is, the economic analysis examined current WTP to avoid development changes that were expected to take place at some time during the next 20 years. Respondents were presented with a set of development alternatives and the expected ecological, economic and social changes that would result from each.

The expected time frame for these effects was not made explicit, making interpretation of the analysis difficult. The time horizon is important both for understanding the respondents' preferences and for comparing the value of current effects to that of future effects (i.e., discounting the stream of future costs and benefits, see Section 2.2.3).[22] Supposing, for example, that respondents assumed most of the expected, high-density development would not occur for 10-15 years in any case – and thus that any benefits of funding an alternative would be similarly delayed – they would have discounted their current WTP accordingly. If development actually is likely to occur sooner than they assumed, WTP values measured in this study would be too small. Similarly, if they assumed that the ecological effects of high density development would

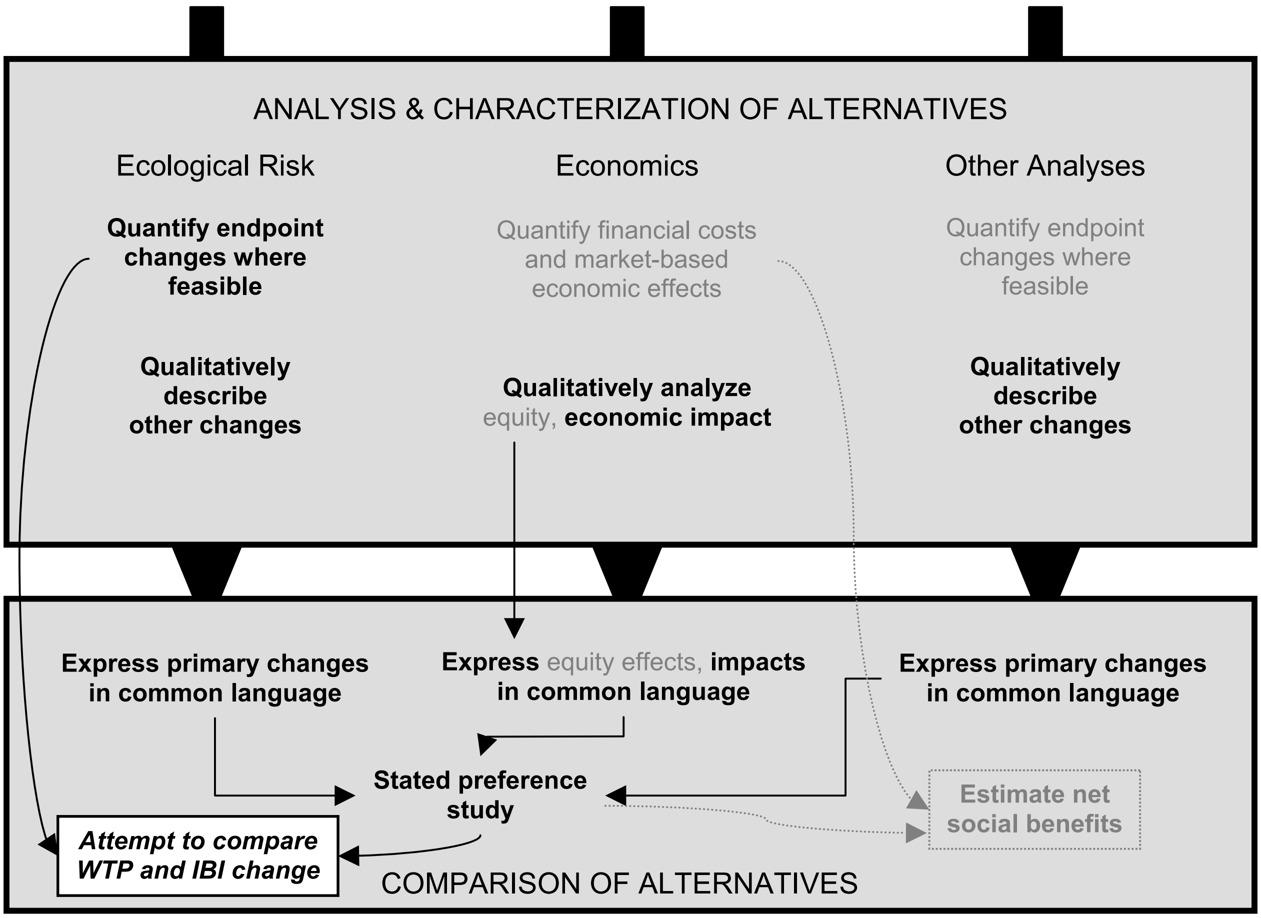

FIGURE 4-3

Techniques used for analysis, characterization and comparison of management alternatives in the Big Darby Creek watershed, as compared to the example shown in Figure 3-3. White boxes and bold type show features included in this analysis.

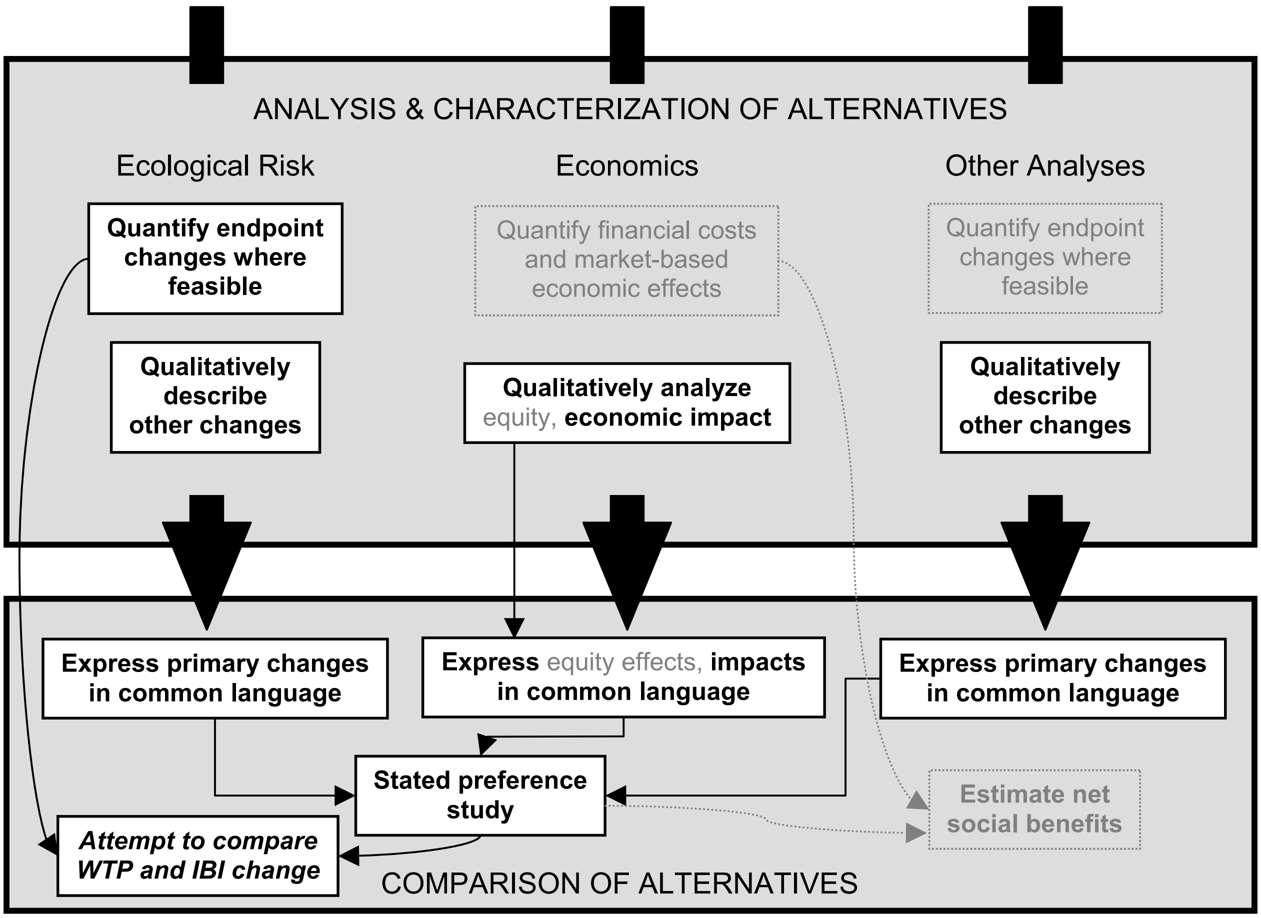

FIGURE 4-3

Techniques used for analysis, characterization and comparison of management alternatives in the Big Darby Creek watershed, as compared to the example shown in Figure 3-3. White boxes and bold type show features included in this analysis.

occur only much later than the other (economic and social) effects, and if this assumption was incorrect, then the ecological benefits of the other scenarios would not matter as much as the other changes and WTP for the more ecologically beneficial scenarios would be negatively biased.

In a subsequent step, WTP was compared to estimated IBI change. This latter step was of limited success, for reasons just discussed in the previous section, but with further analysis it could provide information that is useful in other settings. In general, this integrated assessment process provided decision support only (see Table 3-1); it did not include decisions or subsequent implementation.

In future studies of this type, if estimates of WTP for a given IBI change are sought, a more effective approach might be to elicit preferences for different fish community characteristics and preferences for different housing densities using separate CVM questions (within one survey) or representing these as separate attributes in a conjoint analysis study (see Appendix 2-A). The next step would be to use these data, along with information on the effects of the development scenarios on fish communities and the financial and market-based economic effects of the scenarios, to assess the net social benefits of the scenarios. Such an approach would be less reliant on establishing accurate respondent understanding of the ecological impacts of housing scenario, and it would also allow adjustment for new knowledge about that relationship without repeating the survey. It would also yield a more inclusive indicator (i.e., net social benefit) than WTP alone.

Nonetheless, neither WTP nor net social benefit estimates are necessarily the best endpoint for housing-related decisions in the Darby watershed. In spite of the thoroughness of the biophysical and socioeconomic framing of this CVM study, reviewers of this study at a USEPA workshop held in July 2001 were pessimistic about its likely influence on development decisions in the study area. They cited the substantial private gains to be made by developing individual tracts to the maximum allowable number of housing units, the spatial fragmentation of zoning authorities, and the tendency of zoning boards to respond to the wishes of property owners and developers. In other words, in specific zoning or development decisions there is not an effective mechanism for internalizing the negative externalities of high density development manifested in statewide WTP. There was skepticism that the simple provision of WTP information would make an impact. Although there is some Clean Water Act authority for reducing the water-quality impacts of home construction, road construction and imperviousness, it does not otherwise interfere with local land development.

Although the assessment planning effort that was carried out originally as part of the Big Darby Creek ERA examined a broad suite of watershed problems, the reviewers' observations suggested that this analysis did not adequately characterize the decision context (see Section 2.1.1.1) specific to suburban development. To better determine the applicability of WTP measured in this study to development decisions in the Big Darby Creek watershed, the assessment planning process would need to be revisited. Participants in a renewed process should include members of zoning boards, farm owners, developers, and individuals representing the residents', near-residents' and statewide interests in retaining the ecological, economic and social amenities of the area. They should also include OEPA officials responsible for addressing local stream reach impairments. Interactions could involve the provision of information about these amenities and the impacts of development, discussion of shared values and an attempt to develop consensus goals for this portion of the watershed. Techniques used might include the joint development of future scenarios for the area.[23,24] Further analyses should include development of TMDLs and implementation plans that consider alternative residential (or industrial) development scenarios. Significantly, these plans should include efforts to develop compensation mechanisms whereby those who partially or completely forego development options are compensated, as is done under "transferable development rights" initiatives.

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5. VALUING BIODIVERSITY IN A RURAL VALLEY: CLINCH AND POWELL RIVER WATERSHED

5.1 WATERSHED DESCRIPTION

The Clinch and Powell Rivers originate in mountainous terrain of southwestern Virginia and extend into northeastern Tennessee, flowing into the upper reaches of the Tennessee River (Figure 5-1). The Powell River originally was a tributary of the Clinch River, but both now flow into the upper reach of Norris Lake. The Clinch and Powell River watershed above Norris Lake, also referred to here as the upper Clinch Valley, covers 9,971 km^2 and ranges between 300 and 750 meters in elevation. Historically, it contained one of the most diverse fish and mussel assemblages in North America,[1] yet most of these populations have declined dramatically or been eliminated.[2] The mainstem Tennessee River and many of its tributaries have been dammed, resulting in the loss of habitat for many fish and mussel species, and therefore the upper Clinch and Powell Rivers represent some of the last free-flowing sections of the expansive Tennessee River system. Currently, the Clinch Valley supports more threatened and endangered aquatic species than almost any other basin in North America.[3] Despite implementing recovery plans for most federally protected species in this basin, there is evidence that these species are either declining or becoming extinct at an alarming rate due to impacts from mining, agriculture, urbanization and other stressors.[4]

The Clinch Valley is a traditional rural Appalachian region. The areas are among the poorest in their respective states, with coal mining, agriculture and scattered manufacturing the primary industries. Although the area is very scenic, with a few exceptions tourism is poorly

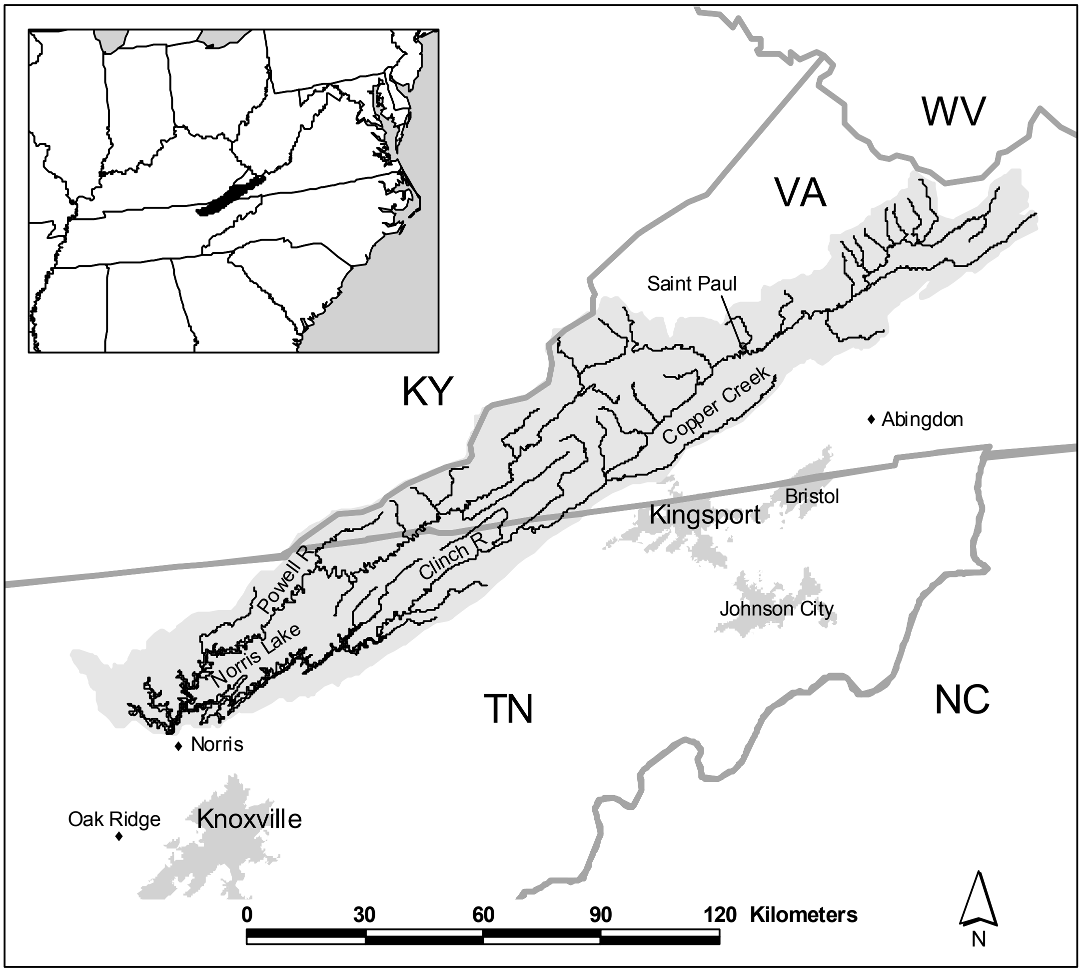

FIGURE 5-1

The Clinch and Powell River watershed in the eastern USA. The study area is the portion of the watershed that is above Norris Lake. Initial ecological study focused on Copper Creek. Towns where discussions were held are shown, as are urbanized areas.

developed. The regional coal and tobacco industries are in decline, and the "high tech" economy has not found its way south of Blacksburg (Virginia Polytechnic Institute and State University) or east of Knoxville (University of Tennessee/Oak Ridge National Laboratory). Many former miners suffer from Black Lung Disease and other problems. School districts often have trouble offering curricula that are comparable to the suburban school districts and finding qualified teachers. Children often leave the region upon completion of their university education.

Transportation problems contribute to the area's economic isolation. Interstates I-81 and I-40 run parallel to the Clinch River, only one or two ridges east, and a quick glance at a map might indicate that transportation is not a problem; however, getting from the Clinch Valley communities to the interstate highways can be quite time consuming, often requiring more than an hour's travel on rural roads. An additional one to two hours is required to reach the Blacksburg/Roanoke area or the Knoxville area. Given the topography of the region, improving the transportation system can conflict with protecting the Clinch River and its tributaries, as the only place for roads is in the flood plains of the streams.

The people of the region do appreciate its environmental resources and are very active in activities such as hunting, fishing and hiking. Evidence of this perspective was found in an unpublished survey. Preliminary to ecological study of the watershed, local environmental organizations surveyed several communities in the region in 1994 to determine their attitudes and values. The results indicated strong interest in protecting local natural resources, but not at the expense of building roads, attracting industry or creating new jobs.

A large amount of ecological information has been collected in this watershed over many years, but much of it had not been analyzed prior to this work. Entities collecting environmental data included The Nature Conservancy (TNC), Tennessee Valley Authority (TVA), U.S. Fish &

Wildlife Service (USFWS), U.S. Geological Survey (USGS), Virginia Department of Game and Inland Fisheries, and Virginia Department of Conservation and Recreation. Resource managers suspected that mining, urbanization and agricultural activities were adversely impacting the exceptional fish and mussel diversity. While several hypotheses have been advanced to explain these species' decline in other watersheds,[5] definitive answers as to their decline in this watershed (Figure 5-2) have been lacking. Resource managers recognized that a comprehensive examination of the available data was needed to evaluate the relative effects of different human activities. Given the socioeconomic context of the Clinch Valley, it is also important to investigate the ways the people of the region compare environmental protection with economic development.

The following sections of this chapter describe studies carried out in the Clinch Valley by the U.S. Environmental Protection Agency (USEPA) and its partners to improve management of the areas unique ecological resources. Section 5.2 describes a watershed ecological risk assessment (W-ERA), initiated in 1993 and carried out by an interagency workgroup. In 1999, USEPA awarded a grant to the University of Tennessee for an economic study that would use the results of the W-ERA and address decision-making needs; this study is described in Section 5.3. Section 5.4 then examines the overall work in the light of a conceptual approach for ERA-economic integration in watersheds (described in Chapter 3).

5.2 ECOLOGICAL RISK ASSESSMENT

5.2.1 Planning

The Clinch Valley ecological risk assessment[7,6,8] was one of five prototype, watershed ecological risk assessments (W-ERA) sponsored by the USEPA to further

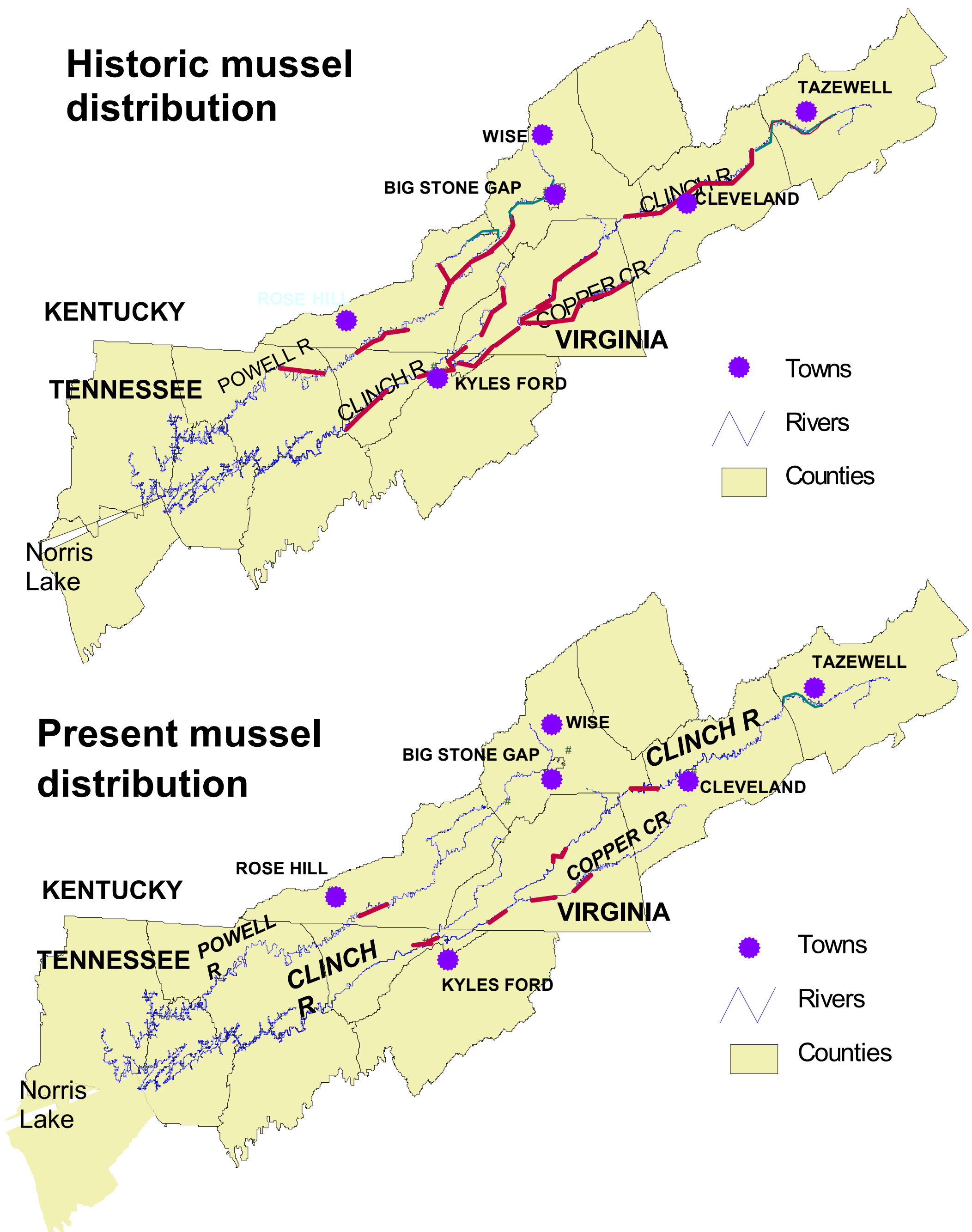

FIGURE 5-2

Comparison between historic (pre-1910) and present locations of native mussel concentrations in the Clinch/Powell watershed; red areas represent mussel beds. (from Diamond et al.[6])

develop, demonstrate and test the use of the ecological risk assessment paradigm[9] at the watershed scale. (The reader is referred to Section 2.1 for more explanation of the procedures and terminology of ERA). Like the other watersheds selected, the Clinch Valley was a candidate for W-ERA because it contains valued and threatened ecological resources, has been the subject of data collection efforts, is subject to multiple physical, chemical and biological stressors and receives attention from several organizations working to protect its resources. Federal, state and local managers had been working with scientists from Virginia and Tennessee to study the distribution of aquatic resources in the Clinch Valley. The global significance of the faunal (especially molluscan) diversity had drawn a great number of scientists to the area.

For this risk assessment, an interdisciplinary, interagency workgroup was established in 1993 with representatives from USFWS, TVA, TNC, Virginia Department of Game and Inland Fisheries, Virginia Cave Board, USEPA and USGS. Unlike in the other W-ERAs, a broader stakeholder group was not convened. Information on attitudes and values from the community survey mentioned in Section 5.1 was taken in lieu of direct stakeholder involvement. Among six environmental concerns presented in that survey, "preserving our rare plant and animal species" was rated lowest in importance, whereas "our water quality" was rated highest. This information stood in some contrast to the urgency for biodiversity protection felt by members of the interagency workgroup.

To focus the scientific information that would be analyzed in the Clinch/Powell watershed, the workgroup identified outstanding ecological resources, developed a management goal and identified a set of management objectives considered important to achieving the management goal (Table 5-1). The workgroup agreed to focus the assessment on the

TABLE 5-1 Outstanding ecological resources, environmental management goal and management objectives for the Clinch Valley ecological risk assessment
Outstanding ecological resources: • *The diversity and biological integrity of aquatic macroinvertebrates, especially the unique native freshwater mussels* • *The diversity and abundance of the native fish community*
Environmental management goal and subgoals: *Establish and maintain the biological integrity of the Clinch/Powell watershed surface and subsurface aquatic ecosystem.* • Establish self-sustaining native populations of macroinvertebrates and fish • Improve water quality in the rivers • Establish and maintain functional riparian corridors of native vegetation • Safeguard water quality in a sustainable sub-surface ecosystem
Management objectives: • Create and maintain vegetated riparian zones in agricultural areas to intercept sediment, nutrient, and pesticide runoff; enhance fish habitat; reduce thermal stress in smaller headwater streams; and exclude cattle from stream beds • Create and maintain vegetated riparian zones in urban, industrial, and developed areas to diminish sedimentation from storm water runoff and reduce instream habitat alteration • Implement agricultural best management practices (BMPs) such as rotational grazing to reduce sedimentation, pathogens, and nutrient enrichment instream • Contain and treat runoff from mining activities to reduce pollutant load and sedimentation instream • Install or improve sewage treatment facilities in streamside rural and urban communities to reduce inputs of toxic pollutants, pathogens, and nutrients instream • Adequately treat industrial discharges to reduce input of toxic pollutants instream • Create and maintain storm water retardation and holding facilities for highways and developed areas to reduce sedimentation runoff instream

From Diamond et al.[8] and USEPA[10]

unimpounded stream segment above Norris Lake, since only that portion of the watershed provided suitable habitat for the fish and mussel species of concern. The assessment would use its limited funds to analyze data collected previously. Terrestrial and aquatic communities in caves associated with karst, though unique and diverse in the watershed, were not examined in this risk assessment because of insufficient information. The workgroup also recognized that there were other possible sources of stress in the watershed, including competition from exotic species (e.g., the asiatic clam *Corbicula fluminea*) and atmospheric deposition of contaminants. They opted not to consider these sources in this assessment because their impacts are relatively minor and they cannot be addressed by local managers.

5.2.2 Problem formulation

During problem formulation, the broad management goal of establishing and maintaining biological integrity was more explicitly defined. Human-caused sources and stressors in the watershed were listed (Table 5-2) and considered in detail.[6] Assessment endpoints corresponding to the outstanding biological resources were selected, and conceptual models were drawn illustrating the pathways by which the endpoints may experience adverse effects. The two endpoints selected in this assessment were: (1) reproduction and recruitment of threatened, endangered or rare native freshwater mussels; and (2) reproduction and recruitment of native, threatened, endangered or rare fish species.

Conceptual models developed by the workgroup traced the most important, hypothesized pathways between sources, stressors, and direct and indirect ecological effects. For example, the model for effects on mussels (Figure 5-3) shows agriculture, mining, silviculture and urban areas to be sources of excess sediment. The resulting turbidity affects mussel survival and recruitment

TABLE 5-2

Stressors and sources identified in the Clinch and Powell watershed

Stressor	Sources	
Degraded Water Quality		
Toxic chemicals	Catastrophic spills Urbanization Point-source discharges Atmospheric deposition	Agriculture Coal mining Transportation
Pathogens	Urbanization	Agriculture
Nutrients	Urbanization Atmospheric deposition	Agriculture
Physical Habitat Alteration		
Sedimentation	Coal mining Hydrologic changes Transportation	Agriculture Urbanization
Riparian modification	Agriculture Hydrologic changes	Urbanization
Instream destruction	Agriculture Hydrologic changes	Urbanization
Biotic Interactions		
Exotic species introductions	Accidental (Asiatic clam, zebra mussel) Recreational (brown trout, rainbow trout)	
Overexploitation	Other biota Over harvesting	Poaching

From Diamond et al.[8]

Conceptual Risk Model for Mussels

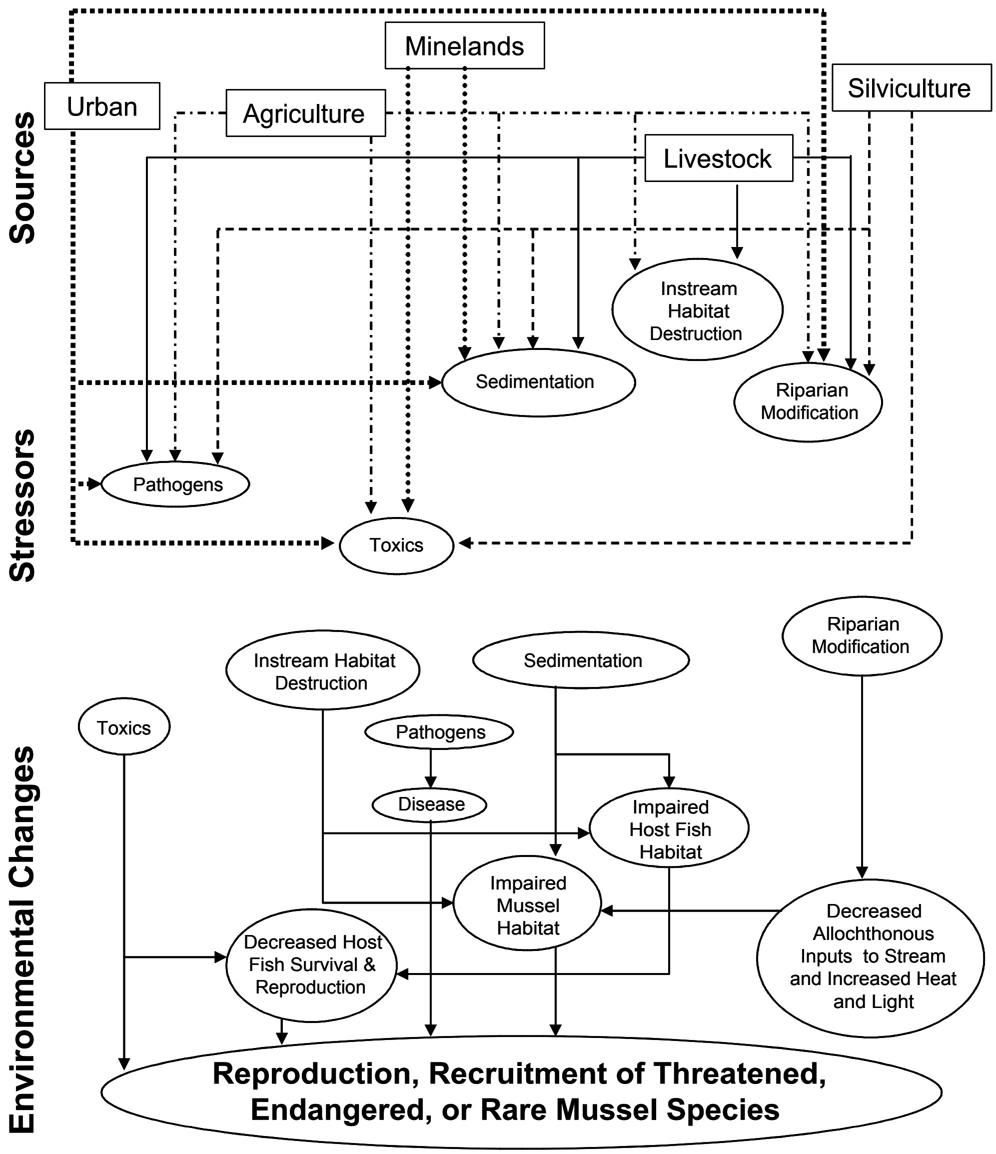

FIGURE 5-3

Simplified conceptual model showing major pathways between sources (land use), stressors, and effects on the assessment endpoint for native mussel species abundance and distribution and data sources available (adapted from Diamond et al.[8]).

by interfering with filter feeding, and siltation smothers the substrates to which they attach. Siltation also smothers benthic (bottom-dwelling) macroinvertebrates, the food source of insectivorous fish, thereby reducing the availability of host species for the mussels' parasitic larval stage, or glochidia, which must attach onto the fins, epidermis or gills of a suitable host fish. A similar model (not shown) traced the pathways for risks to fish species.

Risk hypotheses to be evaluated in the analysis phase were developed for each endpoint, and eventually consolidated to three, corresponding to two categories of stressors:

Physical Habitat Alteration Hypotheses

- Greater connectivity of riparian (i.e., stream-side) vegetation, or forested riparian vegetation, is associated with greater diversity and abundance of mussels, other macroinvertebrates, and native fish.
- Watershed areas dominated by agricultural, urban, or mining land uses are associated with poorer physical habitat quality and biological diversity than are forested or naturally vegetated areas.

Water Quality Hypothesis

- Proximity to nonpoint-source runoff (from agricultural activities and urban areas) and point-source discharges (including coal mining discharges) results in detrimental structural changes to native mussel and fish populations.

Available data sets for subwatersheds of the Clinch Valley were examined and an analysis plan was developed. Because of data limitations, it was decided to undertake a preliminary analysis in a subwatershed, Copper Creek (Figure 5-1), to determine the appropriate spatial scale for analysis of riparian vegetation and land uses, and to identify appropriate biological measures as surrogates for the assessment endpoints. It was also decided that TVA would organize the available information in a geographic information system (GIS).

5.2.3 Risk analysis

5.2.3.1 Methods

Analyses were based on data collected at many locations in the watershed over several years. Monitoring programs that provided key data for this risk assessment included TVA's Clinch-Powell River Action Team Survey and the Cumberlandian Mollusc Conservation Program. Land cover data used in this risk assessment were derived from LANDSAT Thematic Mapper imagery, classified into 17 discrete categories including several different forest types, urban and developed land, pasture and cropland. All terrain data (e.g. elevation and slope) were derived from a mosaic of USGS digital elevation models (DEM) at 30-m resolution. USEPA's River Reach File 3 provided stream network data. Locational data were also available for mines, coal preparation plants, major transportation corridors, urban centers, and biological sites in the basin. Several measures of instream habitat quality, including bottom substrate characteristics, bank stability, riparian vegetation integrity, channel morphology and instream cover, were used to characterize habitat condition. A multimetric habitat quality index (similar to QHEI; see Appendix 2-B) was also used. However, water quality data were insufficient to allow determinations either of land-use effects on water quality or water-quality effects on the assessment endpoints. Therefore, it was necessary to directly examine the relationships between land uses, instream habitat quality and the assessment endpoints, without reference to water quality *per se*.

Since data directly matching the assessment endpoints were not available, surrogate measures were used. For example, few data were available on native threatened, endangered or rare fish species. However, the Index of Biotic Integrity (IBI), a multimetric index describing the status of the fish community, had been determined by TVA at a number of locations throughout the watershed and was considered to be a reasonable measure for the second assessment endpoint (for more information on the IBI see Appendix 2-B). Data on mussel species richness and abundances were also limited, but preliminary study in the Copper Creek subwatershed showed a reasonable correlation between IBI score and mussel species richness, and therefore IBI values were used to supplement the mussel species data.[7] For benthic macroinvertebrates, the EPT index, consisting of the number of taxonomic families present from the orders Ephemoptera (mayfly), Plecoptera (stonefly) and Tricoptera (caddisfly), had been determined in some locations. These orders are known to be sensitive to adverse water quality and are replaced by other macroinvertebrates as water quality diminishes.

Forward stepwise multiple regression analyses and/or univariate statistical analyses of data within a GIS were used to test stressor-response associations. GIS maps were produced that examined each risk hypothesis. In many cases it was necessary to reduce the underlying variability by truncating the elevation range of sites included in order to detect source-response or stress-response relationships.

5.2.3.2 Copper Creek pilot study

Copper Creek was chosen for pilot analysis because it was a comparatively data-rich subwatershed, and it presented a simpler case in that agricultural uses were the major sources of stressors. Findings, which were used to structure the analysis of the entire Clinch Valley watershed, included the following:

- Agricultural uses in the riparian zone had more of an influence on instream habitat quality and fish community integrity (IBI) than did upland agricultural land use

- Effects of human activity in the riparian zone could be observed in native fish and mussels as much as 1500 m downstream of the activity (e.g., Figure 5-4)
- IBI score was correlated with mussel species richness
- Land use in the riparian corridor had a stronger effect on IBI than did an overall index of habitat quality, although particular habitat parameters – such as instream cover score, and degree to which stream substrates were free from embedding fine sediments (clean substrate score[a]) – did correlate well to IBI and EPT
- After analyzing riparian corridor data at widths of 50, 100, 200 and 500 m and at varying lengths, a riparian corridor zone measuring 200 m across (100 m to either side of the stream) and extending 500 to 1500 m upstream was found to be the appropriate spatial area in which to analyze land-use effects on fish and mussels.

5.2.3.3 Clinch Valley

The most successful analytical approaches in the Copper Creek pilot study, noted above, were applied to the entire Clinch Valley watershed. Because other parts of the watershed are subjected to stressors from the coal industry and urbanization, the riparian land cover analyses were expanded to include land uses other than agriculture. Land use analyses included the following:

- Proximity to different types of mining activities
- Proximity to urban/industrial areas
- The percentage of land use in the area that was forested, pasture, cropland, or urban

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[a] TVA defines this parameter as “substrate embeddedness.” To make the directionality of the score (1 = poorest, 4= best) more intuitive, it is here renamed “clean substrate score.

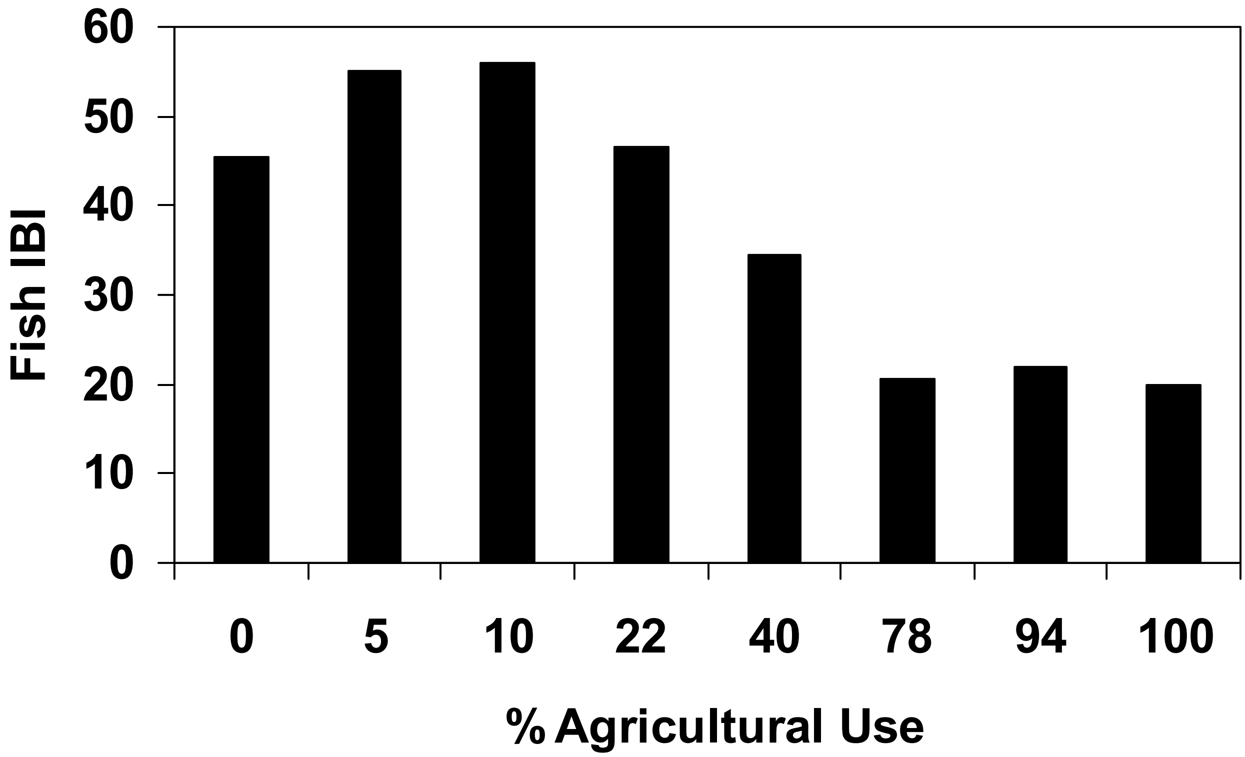

FIGURE 5-4

Fish community integrity as a function of agricultural land in a riparian corridor of 200 m width and 1500 m length in Copper Creek (from Diamond et al.[8])

- Proximity to three classes of roads, including major U.S. highways, State roads, and county roads.

5.2.3.3.1 Effects of land use on habitat quality

Some effects of riparian-corridor land use upon instream habitat quality could be discerned when variability was reduced by limiting sites analyzed to those occurring between 350 and 450 m elevation. Forty-two percent of among-site variability in the habitat quality index (N = 85) could be explained by riparian land use. Stream sedimentation was lower where cropland was 3% of total land use. Riparian integrity was better in areas in which pasture or herbaceous land was < 50% of the total land use. Instream cover was poor if urban use was 20% of the surrounding area upstream. Instream cover and clean substrate scores were affected by both percent pasture/herbaceous cover and percent urban area nearby. The relationships between land use and habitat quality suggest that instream habitat will have the highest probability of being satisfactory for aquatic life if agricultural land use is relatively low and urban influences are small.

5.2.3.3.2 Relationships between land use and biological measures of effect

Among sites of 350 - 450 m elevation, riparian land uses explained 55% of variability in IBI scores (N = 38) and 29% in EPT scores (N = 34). Percent pasture area was positively related to IBI while proximities to mining, crops and urban areas were negatively related. The apparently positive effect of pasture land on IBI was unexpected based on the pilot results for Copper Creek and the negative relationship between pasture area and riparian integrity observed at these sties. A likely explanation is that IBI may respond positively to moderate nutrient enrichment and that negative effects of mining and urban development are comparatively much worse. The number of native mussel species was inversely related to several land uses including (in order of significance): percent urban area; proximity to mining; and percent cropland. In the multiple regression model these factors accounted for 26% of the observed variation in mussel species richness. Collectively, the analyses demonstrated that mining and urban areas are more detrimental than pasture areas to aquatic fauna in this watershed.

5.2.3.3.3 Relationships between habitat quality and biological measures of effect

In stepwise regression analyses of sites 350 - 500 m in elevation, habitat measures proved less effective than land uses at explaining variance in biological measures. Regression models explained 29% of the variance in IBI (N = 81) and 23% in EPT (N = 65). However, in univariate analyses where IBI was categorized as either poor or good based on TVA's criteria, both instream cover and clean substrate scores were clearly related to fish IBI: sites with either low instream cover or highly embedded substrates had a >90% chance of having poor fish community integrity (Figure 5-5). The low overall explanatory power indicates either that both of these biological measures were responding primarily to non-habitat related factors or that the habitat quality measures used were not sufficiently sensitive indicators of physical stressors in this basin.

5.2.3.3.4 Cumulative source index for each site

A cumulative source index for each site was computed, based on how many of four stress-causing land uses (sources of stressors) were present within 2 km upstream of the site. The four sources were: active coal mining or processing; major transportation corridors; > 10% urban area; and > 10% cropland area. IBI was inversely related to the cumulative number of sources present (Figure 5-6A) and was consistently "poor" or "very poor" (TVA rating) at sites

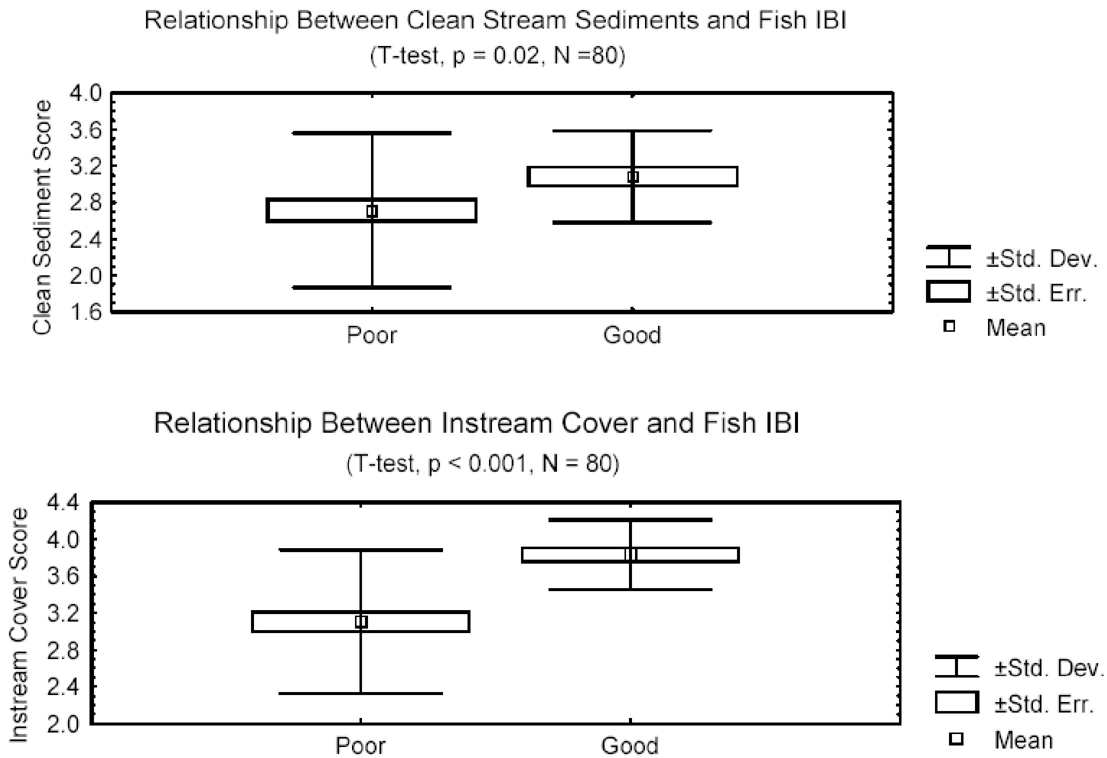

FIGURE 5-5

Relationship between two instream physical habitat parameters, clean sediment (substrate embeddedness) and instream cover, and IBI score, where IBI is categorized as either poor (impaired) or good (unimpaired) based on TVA's criteria; fish community impairment is associated with poorer habitat quality as measured by these two parameters (from Diamond et al.[6]).

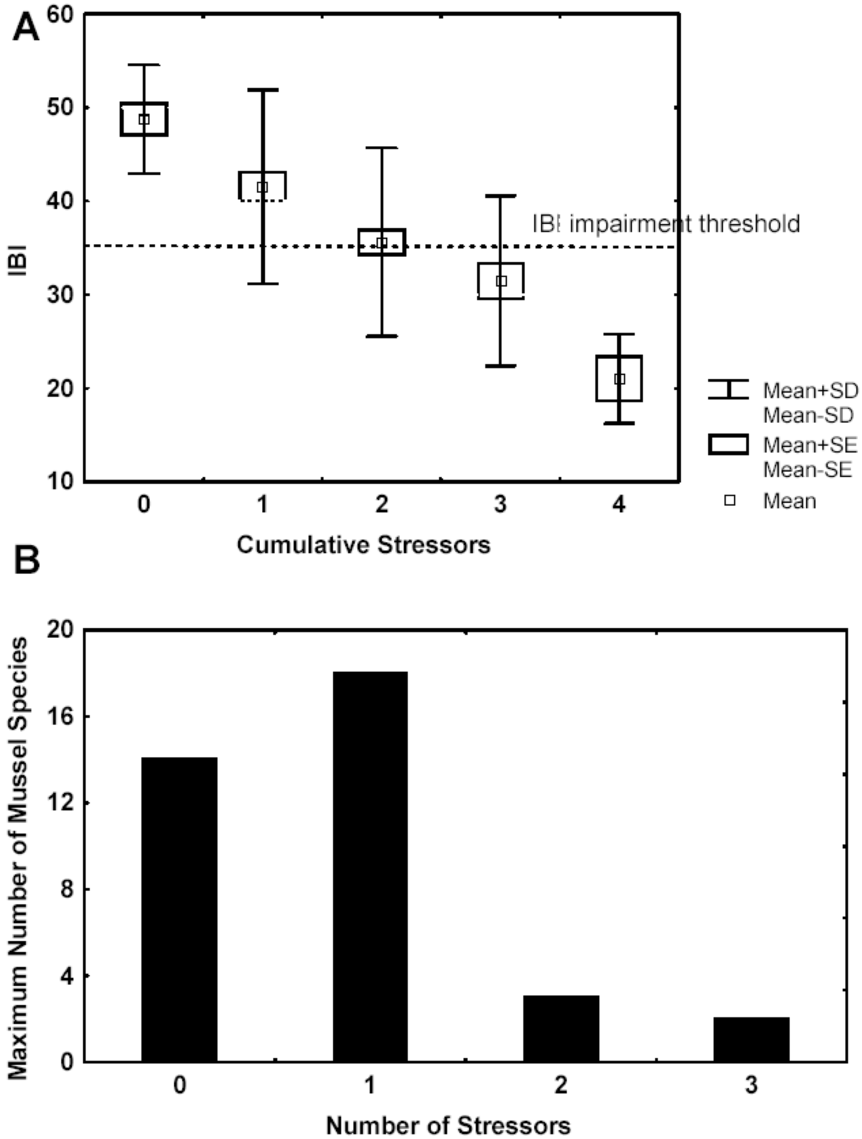

FIGURE 5-6

Fish IBI (A) and maximum number of mussel species (B) in the Clinch/Powell basin as a function of the number of stressors (from Diamond and Serveiss[6]) having all four sources present. In nearly all of these cases (88%), the proximal sources were urban areas and mining. Similar results were found for the maximum number of mussel species present at a site (Figure 5-6B). Sites having 2 or more proximal sources had a >90% probability of having fewer than 2 mussel species present. Sites with one or no sources of stress had between 4 and 18 species, which is still far less than the historical number of species reported (>35 species at many sites[11]).

5.2.3.3.5 Potential effects of toxic chemicals

The risk analysis was hampered by the lack of water quality data sufficient for examining correlations between water quality parameters, including toxic chemical concentrations, and biological effects. The significant amount of variance in biological indices that was unexplained by land use and habitat quality data suggests that other factors were at play. Toxic chemicals may be released in municipal or industrial effluents, from coal mining or processing activities, or transportation accidents. While macroinvertebrates can recolonize an area within a relatively brief period following an episodic release, recolonization by fish and especially molluscs may require years or decades, depending on distance and barriers to other colonized areas. Figure 5-7 illustrates effects observed after catastrophic spills at Westmoreland Coal Company and the APCO power plant on the Powell and Clinch rivers, respectively. In 1998, a large coal slurry impoundment on the upper Powell River failed, resulting in a massive fish kill and substantial mortality of native mussels for a distance of more than 20 miles downstream. A 1999 truck accident on the upper Clinch River in the Cedar Creek area resulted in substantial loss of mussels, including more than 300 threatened and endangered mussels.[12]

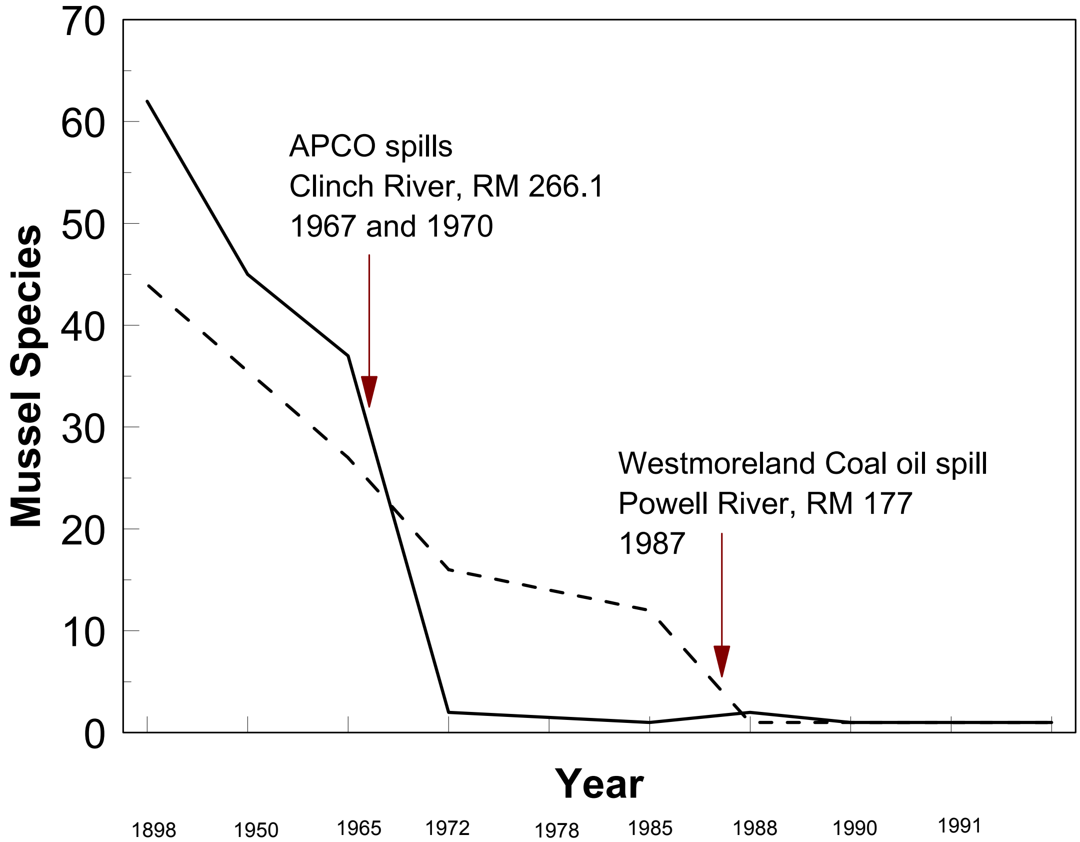

FIGURE 5-7

Number of mussel species recorded over time at two sites in Clinch/Powell watershed affected by large toxic point-source discharge events (from Diamond et al.[6])

5.2.4 Risk characterization

Risk analysis examined the available data on land use, instream habitat parameters and biological assemblages and produced a limited set of statistical associations. The risk characterization step interpreted these associations to suggest what the primary sources of risks are and to explain observed trends in stream faunal diversity. It also described uncertainties and presented management recommendations.

5.2.4.1 Ecological risks

Analyses indicated that up to 55% of the variability in stream fauna could be explained by land uses, with mining and urban land uses exerting the most adverse effects. Key factors appeared to be sedimentation and other forms of habitat degradation stemming from urban and agricultural land uses and toxics from coal and urban areas. Riparian areas with more forested land cover and less cropland, urban, or mining activity tended to be associated with less sedimentation, more instream cover for aquatic fauna, cleaner substrates, and higher fish and native mussel species richness. Our results suggest that if agricultural or urban use upstream is great enough within the riparian zone, sedimentation effects and subsequent loss of habitat will ensue for some distance downstream (1-2 km). These effects are accentuated in higher-gradient, headwater areas.

Although riparian vegetation can reduce deleterious land use effects on water quality,[13] it is not clear that improvement of the riparian corridor alone in this watershed will necessarily result in recovery of native mussel and fish populations. Little or no recovery of threatened or endangered mussel or fish species has been observed in this basin despite improved water quality.[1] In fact, results of this study suggest that the risk of native species extirpation is likely to increase as more sources of potential stress co-occur. Of 10 remaining mussel concentration sites studied, only half appeared to be reasonably isolated from major roads, urban areas, mines, and agricultural areas. This information suggests that native mussel populations are relatively vulnerable to likely sources of stress in this watershed and that further extinctions or extirpations are probable unless additional resource protection measures are taken.

Native fish and mussels have a high risk of extirpation due to endemism (i.e., restriction to a very limited geographic area) and habitat fragmentation, resulting in populations that are too inbred, small in size, and more susceptible to stressors. Populations are now more widely separated than they were historically,[3] which could lead to reduced recruitment success and declining populations, especially in the presence of stressors. Therefore, it may be most useful to further protect those populations that appear vulnerable due to proximity to mining, urban areas, or transportation corridors. Protection and/or enhancement of the riparian corridor at these sites, as well as protection from toxic spills and discharges, is probably as important for sustaining endemic species as stocking new or historically important areas. If stream habitat as well as water quality can be maintained or improved, present mussel and fish populations might be able to expand into nearby areas, thus increasing the distribution and abundance of these species.

5.2.4.2 Uncertainties

Several uncertainties limited our ability to discern associations between causes and effects in the upper Clinch Valley. First and foremost, as has just been noted, the available biological information was only infrequently coincident in time and place with relevant instream chemical measurements. Second, physical habitat assessment data were fairly qualitative and relatively infrequent. Given the observed importance of physical stressors such as sedimentation on valued resources in this water body, resource managers should use more robust habitat assessment techniques that provide more quantitative data on impairments. Third, the macroinvertebrate measure EPT relies on family-level taxonomy, reducing its ability to discriminate changes in the benthic community; a generic- or specific-level index probably would provide better information. Fish IBI appeared to be a more sensitive index to stressors, probably because the metrics in this index have been demonstrated to be sensitive in a number of other watersheds. Fourth, the apparent relationship between fish IBI and mussel species richness or abundance, observed in the Copper Creek subwatershed, needs to be explored in more detail. IBI is composed of a number of metrics, such as native species richness, that were potentially more explanatory of mussel assemblages, but the unaggregated data were not available to this analysis. It must be noted, however, that any comparisons between native mussel and fish or macroinvertebrate data will be limited by the lack of overlap in sampling locations between TVA's monitoring programs. Only eight sites in the entire watershed had data on mussels and either IBI or EPT. Because of the paucity of mussel species occurrence data, the risks to mussel species in the watershed could be over- or understated.

5.2.4.3 Management recommendations

The risk assessment has helped lend further credence to what many resource managers had long conjectured were problems within the watershed, thereby providing more scientific support to take actions to address problems. Based on the assessment findings, the USFWS and TNC are considering the following types of management actions: riparian buffer protection; building spill prevention devices along transportation corridors near streams and restricting the type of materials transported over certain bridges; limited access of livestock to streams; better monitoring and control of mine discharges to streams; maintaining existing natural vegetation; BMPs for pasture and agricultural land to reduce sediment loading; and better treatment of wastewater discharges.

5.3 ECONOMIC ANALYSIS

The overarching goal for integrated ecological and economic analysis was to utilize the findings of ERA in an economic analysis that would be relevant to environmental management decisions in the watershed. The economists' team chose to focus on values held by valley residents as important for determining how local decision-makers would act. The economic analysis therefore addressed the task of valuing potential changes in biological diversity and other ecological services at risk in the upper Clinch River Valley in Virginia and Tennessee, as expressed by Valley residents. This task presented two major challenges. First, credible measures of economic value needed to be integrated with the ecological assessment endpoints such that the results would be useful in analyzing risk-relevant management and development scenarios. Second, the techniques used in the study needed to be consistent with economic principles of individual welfare maximization and to minimize biases associated with the measurement process.

Ecologists, such as those conducting the W-ERA, and Clinch Valley residents were thought to view the ecological assessment endpoints differently. Ecologists believe that biodiversity is important for a number of reasons, including its contribution to ecosystem resilience, i.e., the ability to withstand perturbations (such as from natural or human-caused stress) without shifting to a different kind of ecological state.[14] As stated earlier, however, Valley residents had rated "preserving our rare plant and animal species" lower than five other environmental concerns listed, and therefore might be unlikely to attach much value to the diversity of the Valley's mussel fauna, for example. However, mussel health is a good indicator of water quality, which residents had rated as most important. Because mussels are very sensitive to pollution, poor water quality will tend to impact mussels before other species in the river, and before human health. The economists expected that Valley residents would value the service provided by mussels as water quality indicators. Their approach was to design a survey that would interpret the results of ERA in terms most likely to be meaningful to Valley residents.

This section is organized as follows: In Section 5.3.1, choice modeling is explored as a potential tool for solving this difficult valuation problem. Section 5.3.2 presents a methodology for integrating a choice modeling approach with ERA in the upper Clinch Valley, and Section 5.3.3 discusses the choice model results.

5.3.1 Methods for valuing biodiversity and environmental quality

5.3.1.1 Conjoint analysis vs. contingent valuation

Current approaches for assessing the value of environmental change, including changes in biodiversity, involve predicting an outcome associated with the change and then using a method such as the contingent valuation method (CVM, see Section 2.2.2 and Appendix 2-A) to estimate individuals' willingness to pay (WTP), for a beneficial change, or willingness to accept (WTA) for a change that is detrimental.[a] For example, Rubin et al.[15] estimate the value of preserving spotted owls in order to determine the benefits of preserving old growth forests, and Stevens et al.[16] calculate WTP for various levels of preservation of Atlantic salmon and bald eagles. However, CVM tends to focus on losing or gaining the whole good, whereas management decisions tend to address changing characteristics of the goods.[17] For example, a typical CVM question might be worded as follows:[b]

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[a] The use of WTP or WTA is a function of the perceived property right as well. See Freeman[27] for a discussion.
[b] This question was contrived for demonstration purposes only. A high-quality CVM survey would convey much more information before the valuation question was posed.

> The upper Clinch/Powell watershed, which lies in southwestern Virginia and northeastern Tennessee, is threatened by water quality insults from agricultural operations, coal processing facilities, and urban runoff. The watershed is important habitat of many plants and animals, including eleven endangered mussels that are found only in the Clinch River. The river and adjacent areas are also used for recreational fishing, canoeing, picnicking, hunting, and to a lesser extent, commercial fishing.
>
> A nonprofit organization is seeking voluntary donations to purchase land and conservation easements to protect water quality in the Clinch/Powell watershed. These lands, which in total would comprise 2,200 acres and would help ensure the protection of 15 miles of stream habitat, would then be managed by state land management agencies as preserved land. Would you be willing to contribute $X to aid in the purchase of the land and conservation easements?

In theory, CVM can measure both use and nonuse components of economic value (see Section 2.2.2); however, all these components would be lumped together in the WTP estimate. By contrast, conjoint analysis (CA) asks individuals to make choices about which state of the world they would prefer, given that different states have differing levels of certain definable attributes. The choice model, a variant of CA, elicits individuals' preferences by asking them to consider a series of trade-offs. In contrast to CVM, which asks individuals to explicitly state their WTP for a proposed change in environmental quality, choice models ask individuals to choose from a series of possible outcomes (choice sets). This allows the researcher to obtain the trade-offs that an individual is willing to make between any attributes presented in the choice sets, as well as to estimate WTP.

Choice models ask questions that may be more familiar to individuals. Individuals are asked to choose among bundles of goods according to the level of attributes of each bundle. For example, individuals routinely make choices among goods that have multiple attributes, such as among five automobiles having different colors, engines, interiors, etc. A typical choice task might ask the subject to choose the most preferred of the five, each having different characteristics, including price. In contrast to CVM, which would ask the individual to assign a price to each of the cars, the choice model task is more representative of the choices that individuals regularly face in making transactions. CA relies less on the information contained in the description of the scenario and more on the description of the attributes of each alternative.[18]

The family of CA models, of which the choice model is a member, is receiving increasing attention in the economics literature as well as in policy circles. Its use has been legitimized by National Oceanic and Atmospheric Administration's (NOAA) proposed Habitat Equivalency ruling, which arose in part due to criticisms of CVM during the Exxon Valdez damage assessment case (60 FR 39816).[a] In particular, NOAA recommended CA as a tool to measure in-kind compensation for damaged natural assets.

Regional development problems and multiple use management are perhaps the ideal tests of the usefulness of the choice model. With proper survey construction, the researcher can measure many characteristics including use and nonuse values, as well as indirect use values such as ecological services (see Section 2.2.2 for definitions of these values). Conjoint models are particularly useful for disentangling likely complementarities between attributes. For

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[a] Habitat equivalency argues that the appropriate measure of natural resource damages due to, say, an oil spill, is provision (or augmentation) of ecological services that substitute for the services lost (e.g., improvement of wetlands in other areas).

example, changes in water quality could be positively correlated with endangered fishes, sport fishing, and water-based recreation; with choice models, the effects of each of the attributes on welfare can be estimated independently.

5.3.1.2 Choice modeling framework

To explain individuals' preferences for alternative states of the Clinch River Valley, this effort used a random utility model (RUM) framework, which is widely used in dichotomous-choice CVM and travel cost modeling, as well as in CA. RUMs rely on choice behavior and assume that individuals will choose the alternative that gives them the highest level of utility; i.e., RUMs estimate the probability that an individual will make a selection based on the attributes and levels of each possible choice. The RUM is directly estimable from choice models (see Appendix 5-B for technical detail of the RUM framework).

5.3.2 Integrating the choice model with the ecological risk assessment

The task of integrating the measurement endpoints from the upper Clinch Valley ERA (especially, IBI and mussel species richness) with indicators of social value proved a formidable challenge, since they were not the type of endpoint the ordinary citizen is likely to think about in his day-to-day life. Meetings were held in Abington, VA and Norris, TN between the economists, ecological risk assessors and other individuals who had shown interest in biological resource management in the Clinch Valley. The decision was made to approach the problem of lack of familiarity with the ecological endpoints in two ways. First, succinct wording was developed to express the relationship of these ecological endpoints to quality of life. After several iterations, a survey was drafted, presented to focus groups, revised and then pilot tested.

Second, socially meaningful endpoints were included that were complementary to the ERA measurement endpoints but outside of the ERA's original scope. For example, increased forestation of the riparian corridor would not only help protect mussel and fish biodiversity but also increase the diversity and abundance of terrestrial fauna and birds and improve the quality of smallmouth bass fishing. Since these endpoints are jointly produced, it was important that they be jointly valued. Their inclusion expanded the choice sets to more fully describe the state of the Clinch Valley environment and the auxiliary benefits of management policies aimed at preserving biodiversity.

5.3.2.1 Choice model design

Choice model surveys are complex by nature. Each possible choice comprises bundles of attributes, with each attribute having different levels. Because the potential for miscommunication between the researcher and the survey recipient via the survey instrument is great, two formal focus groups of 6 and 11 subjects and three informal focus groups were conducted to inform our survey design. The first informal group was conducted in September 2000 using staff and students of the University of Tennessee. The second and more formal focus group was conducted by an expert facilitator in St. Paul, VA in November 2000. The third and fourth focus groups were conducted at the University of Tennessee in January and February 2001. The final focus group was conducted in Oak Ridge, TN in February 2001 using residents of Anderson County, TN, the westernmost county in our study.

The focus groups allowed the participants to home in on those attributes correlated with management changes that are likely to be important to the residents of the Clinch River Valley. Six attributes were identified, with the number of levels per attribute varying from 2 to 6 (Table 5-3); see Table 5-4 for an example choice set from the survey. The “cost to household” attribute allowed the estimation of conventional WTP measures. Interaction with the “Agricultural

TABLE 5-3

Attributes and attribute levels used in survey questionnaire. Attribute levels making up options A and B in a given choice set varied among those listed; attribute levels for option C were the same in all choice sets.[a] Corresponding model variable names are given in parentheses.[b]

<table>
<tr><th>Attribute</th><th colspan="6">Attribute Levels for Options A & B</th><th>Option C: No New Action</th></tr>
<tr><td>Agriculture-free zone</td><td colspan="2">25 yards Clinch /10 yards tributaries (BIGZONE)</td><td colspan="2">10 yards Clinch /5 yards tributaries (SMALLZONE)</td><td colspan="2">none</td><td>none</td></tr>
<tr><td>Aquatic Life</td><td colspan="2">full recovery (FULLRECOV)</td><td colspan="2">partial recovery (PARTRECOV)</td><td colspan="2">continued decline</td><td>continued decline</td></tr>
<tr><td>Sportfish</td><td colspan="2">increase (SPORTINC)</td><td colspan="2">no change</td><td colspan="2">decrease (SPORTDECL)</td><td>no change</td></tr>
<tr><td>Songbirds</td><td colspan="3">increase population (SONGINC)</td><td colspan="3">no change</td><td>no change</td></tr>
<tr><td>Agricultural income</td><td colspan="3">no change</td><td colspan="3">$1 million/yr decrease (AGDECL)</td><td>no change</td></tr>
<tr><td rowspan="2">Cost to Household ($ per year)</td><td>$100</td><td>$75</td><td>$50</td><td>$25</td><td>$10</td><td>$5</td><td rowspan="2">no change</td></tr>
<tr><td colspan="6">(COST)</td></tr>
</table>

[a] The choice sets are designed to allow for the efficient estimation of the parameters of all of the attributes. While SMALLZONE and BIGZONE are our policy variables, they are varied independently of the other variables. For example, it is possible to have choice sets that include the 25yard/10yard agriculture exclusion (BIGZONE), but have SPORTDECL or have CONTINUED DECLINE for the level of aquatic life. Individuals would be expected to focus on the outcomes and not the policy attribute.
[b] See Appendix 5-A for explanatory text that was provided in the survey

TABLE 5-4

Sample question and choice set from survey questionnaire

Which option for the future of agriculture and the environment in the Clinch Valley do you prefer the most, Option A, Option B, or Option C? Option C is the status quo, or what is currently happening and will continue to happen with no further environmental or agricultural policies. Note that some of these options might not seem completely realistic in real life. We ask that you do your best to assume that each option is possible and then choose your most preferred option.

	Option A	Option B	Option C: No New Action
Agriculture-free zone	10 yards Clinch/5 yards tributaries	10 yards Clinch/5 yards tributaries	none
Aquatic Life	full recovery	partial recovery	continued decline
Sportfish	no change	increase	no change
Songbirds	increase	increase	no change
Agricultural income	no change	no change	no change
Cost to Household ($ per year)	$50	$50	no change

Please check the option that you would choose:

Option A ◻ Option B ◻ Option C ◻

Income" attribute allowed investigation of whether individuals think society as a whole, or farmers and ranchers alone, should bear the burden of increased environmental quality.

Choice model variables were specified based on these attribute levels, and *a priori* predictions of their signs were made (Table 5-5). The variables that represent the attributes agriculture-free zone, aquatic life, and sportfish were each decomposed into two separate, effects-coded variables to control for the three levels that each of these variables can take (see Louviere et al.[19] for a full discussion). Effects codes are an alternative to dummy variable codes and are useful when interpreting the coefficients of a choice model.[18,19] SMALLZONE and BIGZONE represent the size of the agriculture-free zone;[a] these are expected to be positive, albeit weakly. PARTRECOV and FULLRECOV should be positive as individuals should be more willing to choose options that lead to higher levels of recovery for aquatic life, other factors being equal. SPORTDECL should be negative as individuals should be less likely to choose options that represent decreases in sportfish populations, whereas SPORTINC should be positive by similar reasoning. SONGINC is expected to be positive, since many people value the presence of songbirds. AGDECL is expected to be weakly negative, since income declines are detrimental to the regional economy but not all respondents expect to be affected directly. COST is expected to be negative; individuals are less willing to choose options that have higher costs associated with them. Alternative-specific constants corresponding to options A and B (ASCA, ASCB) are included to incorporate any variation in the dependent variable that is not

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[a] An omitted third variable for the status quo, NOZONE, is implicit in the model; its coefficient can be determined based on the coefficients of the included variables.

TABLE 5-5

Choice model variables and expected sign

Variable[a]	Expected Influence of Variable
CHOICE[b]	NA
SMALLZONE[c]	+
BIGZONE[c]	+
PARTRECOV[c]	+
FULLRECOV[c]	+
SPORTDECL[c]	-
SPORTINC[c]	+
SONGINC[c]	+
AGDECL[c]	-
COST	-
EDUC	+
AGE	?
RIVERVIS	+
MOSTIMPO	+
FISHLIC	+
ENVORG	+
ASCA[d]	?
ASCB[d]	?

[a] Variable names are explained in Table 5-3 or in text
[b] Dependent variable
[c] Effects-coded variable
[d] Alternative-specific constant

explained by the choice set attributes or respondent characteristics; there was no *a priori* expectation as to their signs.

Selected socioeconomic information thought to be important was also included in the choice model (Table 5-5). For example, it is common (though not universal) in the literature to see more support for measures to improve environmental quality as the level of education increases,[20] so EDUC is expected to be positive. RIVERVIS, which is equal to 1 if the subject visited the Clinch within the last year, is expected to be positive, since individuals are expected to choose outcomes that improve the quality of their visits to the river. Likewise, MOSTIMP (which equals 1 if the individual believes either that recreation is the most important use, or that environmental quality is the biggest issue in the Clinch Valley) is expected to have a positive sign. FISHLIC, which equals 1 if the individual holds a fishing license, should be positive; individuals who fish should be more likely to choose options 1 and 2 that generally include better environmental quality. ENVORG, which equals 1 if the individual belongs to an environmental organization, should be positive. There was no *a priori* expectation about the effect of AGE on choice.

Having defined these parameters, a RUM-based choice model (Appendix 5-B) is developed as follows:

$$\begin{aligned}\text{CHOICE} = {} & \alpha_1\text{ASCA} + \alpha_2\text{ASCB} + \beta_1\text{SMALLZONE} + \beta_2\text{BIGZONE} \\ & + \beta_3\text{PARTRECOV} + \beta_4\text{FULLRECOV} + \text{remaining} \\ & \text{attributes and socioeconomic parameters} + \varepsilon\end{aligned} \qquad 5\text{-}1$$

where the remaining attributes and socioeconomic parameters are all of the remaining terms in Table 5-5.

5.3.2.2 Survey implementation

Final language to describe the choice attributes to respondents was developed (Appendix 5-A). Respondents were asked to answer eight choice sets.[a] An example choice set is found in Table 5-4.

Surveys were mailed to a random sample of 400 households in the Clinch River Valley, with the majority being distributed in the Virginia portion of the valley. [b,c] Principles from Dillman's Total Design Method[21] were followed. Approximately two to three weeks after the survey mailing, a reminder postcard was mailed to thank participants and encourage non-respondents to return their surveys.

5.3.3 Results of economic analysis

Ninety one subjects completed the choice study (response rate was 23%); 76 provided complete responses for all eight choice sets, generating 1824 acceptable observations for analysis (see Table 5-6 for summary statistics).

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[a] A fractional factorial design was employed to develop a survey based on this choice model. A full factorial design would have required 648 (= $3^3*2^2*6^1$) different choice sets. The %MKTDES macro in SAS was used to choose 16 choice sets that are meaningful and will still allow the main and interaction effects to be estimated. These 16 choice sets were then blocked into two blocks of eight choice comparisons. One outcome of the focus group process was that subjects indicated that the 16 choice sets that they had initially evaluated were too many.

[b] The delivery envelope for the survey was personalized and included a cover letter, the survey, supporting documents, and a stamped return envelope. Surveys were printed on legal size (8.5"x14") paper folded as a 20-page booklet and stapled along the spine. The supporting documents were printed on letter size paper.

[c] This survey was distributed as part of a larger study employing four different survey versions. The other surveys allowed the examination of the trade-offs of strictly environmental attributes such that a preference-based index can be constructed; a version where mussel protection implies trade-offs in employment in several sectors of the economy; and a version designed to test the similarities between choice and contingent valuation models. Results of the other surveys are still pending.

TABLE 5-6

Summary statistics

Variable	Mean	Std. Dev	Min	Max	Observations[a]
EDUC	13.409	1.426	6	16	1800
AGE	45.855	14.723	18	81	1824
RIVERVIS	0.592	0.492	0	1	1824
MOSTIMPO	0.627	0.484	0	1	1800
CHOICE	0.333	0.472	0	1	1824
SMALLZONE	-0.249	0.8239	-1	1	1824
BIGZONE	-0.236	0.836	-1	1	1824
PARTRECOV	-0.101	0.902	-1	1	1824
FULLRECOV	-0.287	0.746	-1	1	1824
SPORTDECL	-0.476	0.707	-1	1	1824
SPORTINC	-0.328	0.875	-1	1	1824
SONGINC	-0.157	0.987	-1	1	1824
AGDECL	-0.358	0.933	-1	1	1765
COST	24.391	31.810	0	100	1824
FISHLIC	0.453	0.498	0	1	1800
ENVORG	0.200	0.400	0	1	1800

[a] There are 1824 possible observations representing 3 possible choices on 8 choice occasions for each of 76 subjects. Ninety one subjects completed this version of the choice study, but only 76 have complete responses for all eight choice sets.

5.3.3.1 Results of choice model estimation

The interpretation of the coefficients in conditional logit models suggests how utility or satisfaction changes given a change in the attribute. The parameters also reveal how the probability that an alternative is chosen changes as the level of the attribute changes.

Parameter values obtained for the discrete choice model generally show the expected signs and the joint power of the model is very good, as evidenced by a McFadden's R^2 of 0.27 (Table 5-7). The signs of the coefficients on the attribute variables are consistent with the priors. Both small and large agriculture-free zones serve to increase the probability that an alternative is chosen, but the small zone has a stronger effect than was anticipated. Full recovery for aquatic life and increases in sportfish are also positive, whereas decreases in sportfish have a negative effect of the probability of choice. AGDECL is negative and significant, indicating that individuals are less likely to choose alternatives if they know that agriculturalists have to pay part of the costs of recovery efforts. COST is negative and significant, indicating a decreased likelihood of choosing an alternative as the tax price increases.

In this model, each subject generates 24 observations (i.e., 3 possible choices on 8 choice occasions) in the data set; thus, socioeconomic characteristics are invariant across choice sets. The only way to control for socioeconomic effects is through interactions with the alternative specific constants or interaction with the attributes. The decision was made to interact education, age, gender, fishing license, and membership in environmental organizations with the alternative specific constants. The interpretation of these interactions is complicated as well. For example, ASCA*MALE and ASCB*MALE both are negative and significant (Table 5-7), indicating that the probability of choosing Option A or B rather than the status quo in any of the eight choice sets is lower for men than for women.

TABLE 5-7

Results for conditional logit with CHOICE as dependent variable

Variable	Coeff	Std. Error	T-statistic	P-value
SMALLZONE	0.697	0.155	4.497	0.000
BIGZONE	0.306	0.158	1.936	0.053
PARTRECOV	0.084	0.148	0.564	0.573
FULLRECOV	0.831	0.150	5.541	0.000
SPORTDECL	-0.727	0.179	-4.054	0.000
SPORTINC	0.593	0.127	4.679	0.000
SONGINC	0.079	0.120	0.657	0.511
AGDECL	-0.157	0.069	-2.271	0.023
COST	-0.033	0.004	-8.654	0.000
ASCA	-0.771	1.288	-0.599	0.549
ASCAxEDUC	0.010	0.088	0.119	0.905
ASCAxAGE	-0.013	0.008	-1.508	0.132
ASCAxMALE	-0.624	0.266	-2.345	0.019
ASCAXMOSTIMPO	0.790	0.264	2.993	0.003
ASCAxFISHLIC	0.256	0.250	1.024	0.306
ASCAxENVORG	0.492	0.308	1.597	0.110
ASCB	-1.505	1.465	-1.027	0.304
ASCBxEDUC	0.044	0.099	0.448	0.654
ASCBxAGE	-0.018	0.011	-1.671	0.095
ASCBxMALE	-0.696	0.313	-2.223	0.026
ASCBxMOSTIMPO	0.561	0.310	1.806	0.071
ASCBxFISHLIC	0.700	0.302	2.318	0.020
ASCBxENVORG	0.246	0.379	0.648	0.517
Number of Observations[a]	526			
Log-Likelihood	-423.759			
Log-Likelihood(0)	-577.870			
McFadden's Rho-square	0.267			

[a] There are 608 choice occasions in the data set, but only 526 observations have complete responses for the variables in the regression. A choice occasion represents a set of three alternatives: one outcome is selected as the preferred option by the individual, the other two are not.

5.3.3.2 Calculation of part-worths

Using the coefficients from Table 5-7, implicit prices (with respect to the COST variable) were obtained for each of the choice variables (Table 5-8). These are typically called the part-worths in the conjoint/choice model literature.[a] While in theory the calculation can be made in terms of any one attribute for any other, the most intuitive trade-offs are those between dollars and the other attributes. We can estimate the part-worths by dividing the coefficient on one of the attribute variables by the coefficient on the COST variable and multiplying that result by negative 1. For example, the part-worth on full recovery of aquatic life is

$$\text{Dollar value of full recovery of aquatic life} = -\left(\frac{\beta_4}{\beta_\$}\right) \qquad \text{5-2}$$

where $\beta_\$$ is the coefficient on the variable COST. Respondents were willing to pay substantially more for a small than for a large agriculture-free zone, suggesting perhaps that (a) the idea of an agriculture free zone is attractive in and of itself, independent of any benefits expressed in the other attributes, but that (b) such land use restrictions are most attractive when kept to a minimum. The dollar-valued part-worth for partial recovery of aquatic life was insubstantial in comparison to full recovery, and that for an increase in songbirds was similarly insubstantial in comparison to that for improved sport fishing, or to the negative part-worth associated with a decline in sport fishing.

[a] This is the marginal rate of substitution concept in economics upon which indifference curves are based. Simply, it gives the trade-offs that an individual is willing to make between bundles of goods while holding utility constant.

TABLE 5-8

Implicit prices, or implied willingness to pay for a given attribute level as compared with the status quo

Attribute	Implicit price ($)[a]
SMALLZONE	21.12
BIGZONE	9.27[b]
PARTRECOV	2.55[c]
FULLRECOV	25.18
SPORTDECL	-22.03
SPORTINC	17.97
SONGINC	2.39[c]
AGDECL	-4.76

[a] Since the payment vehicle described in the survey was a change in tax rate (see Appendix 5-A), values should be assumed to represent annual amounts.
[b] Coefficient for BIGZONE was marginally significant (see Table 5-7)
[c] Not significantly different from zero

5.3.3.3 Calculating the value of a biodiversity management program

Economists are often interested in calculating the change in welfare, or well-being (Section 2.2.1), due to a change in public policy. The β estimates allow the calculation of compensating variation (CV), or total WTP, associated with any policy definable in terms of the attributes. First, the utility of the status quo is calculated by substituting the appropriate variable values defining the status quo attribute levels into Equation 5.1. Next, the utility of the policy is calculated using the values corresponding to the attribute levels that define the policy. Then, CV is given by

$$\mathrm{CV} = -\frac{1}{\beta_{\$}}(\textit{status quo utility} - \textit{utility of new policy}) \qquad \text{5-3}$$

Following these techniques for obtaining CV[22] and using the coefficients in Table 5-7, the choice model allows valuation of the multi-attribute change to be evaluated (e.g., in the case where management actions lead to simultaneous improvements [or declines] in the various facets of the ecosystem). If, for example, the status quo utility were taken as zero and a change in agricultural practices were to improve habitat for mussel populations, sportfish, and songbirds–and farmers' income were unaffected by the program–the welfare for the representative individual would increase by $54.81 (i.e., the average respondent would be willing to pay $54.81 annually to move from the status quo to the state of the world having the new agricultural practices). It is this ability to derive multiple welfare measures for complex ecosystem changes that sets choice models apart from CVM studies that allow calculation of the value for only a single policy change.

5.4 DISCUSSION

This section evaluates the cumulative outcome of the W-ERA and economic analysis conducted in the upper Clinch Valley by comparison to the generalized conceptual approach for ERA-economic integration developed in Chapter 3 (see Figure 3-1). As explained in Section 1.5.1, the Clinch Valley analyses were undertaken prior to the development of this conceptual approach, and the economic analysis was initiated following completion of the W-ERA. For these reasons, the studies conducted in the Clinch Valley should not be viewed as integrated in any ideal sense. However, the conceptual approach for integration can be used to examine these efforts in the larger context of watershed decision-making and management and to gain insights as to ways that integration can be improved. The following discussion compares specific components of the conceptual approach with work carried out in the Clinch Valley.

5.4.1 Consultation with extended peer community

The conceptual approach for integration has defined the "extended peer community" as consisting of interested and affected parties, decision-makers, and scientific peers and has argued, in agreement with the National Research Council[23] and others,[24-26] that these parties should be actively engaged throughout assessment processes (see Sections 2.1.1.5, 3.2 and 3.3.5). The ERA for the upper Clinch Valley was undertaken by a diverse, interdisciplinary and multiagency workgroup that included both government and nongovernment representatives, and the risk characterization was conducted with scientific consultation (a workshop held by USEPA) and formal peer review. The result was a creative, state-of-the-art analysis, the findings of which have helped to identify potential management actions by workgroup member organizations.

Decisions were made at an early stage to conduct the W-ERA without an open process for broader, public involvement. Through an informal survey, and long experience working in the region, analysts had indications that community residents valued biodiversity less highly than water quality, on one hand, and economic development opportunities, on the other. Therefore, the management goal on which the ERA was based, which focused on biological integrity, reflected the values of the technical specialists and environmental managers who composed the interagency workgroup, rather than a broader stakeholder consensus as in other W-ERAs. This decision undoubtedly allowed the workgroup to tackle the difficult problems of data gathering and analysis more expeditiously; arguably, it may also have limited the development of broader community awareness of biodiversity issues and mutual understanding of necessary trade-offs for environmental protection.

The economic analysis team benefited from several consultations with members of the ERA workgroup and selected stakeholder group representatives, in which ERA findings were explained and regional economic development goals were discussed. Informal and formal consultations (focus groups) with watershed residents were held to avoid miscommunication between analysts and the public. The resulting survey instrument may be thought of as a structured form of consultation with the public, in which aspects of ecological risk were presented and feedback, in the form of choices between alternative states, was elicited. Interestingly, certain results of the economic analysis ran counter to expectation about residents' values. Survey respondents appeared willing to trade-off a portion of regional agricultural income in order to obtain full recovery of aquatic life, and willing to accept—even to help fund—measures that would limit agricultural use of the riparian zone to improve habitat.

5.4.2 Baseline risk assessment

The conceptual approach for integration defines baseline risk assessment as the assessment of risks currently and into the future if no new management action is taken (Section 3.3.3). The upper Clinch Valley ERA used existing data to characterize the risks (and uncertainties) affecting the assessment endpoints according to current conditions and trends. It identified the impacts of multiple sources and stressors, and pointed to the future likelihood of continued extirpations of species if stressors are not more effectively managed. It provided models (in this case, empirical relationships) that could be used to assess the impacts of management policies, including spatial relationships of riparian zone land use and in-stream biological response and the impacts of multiple stressors. It did not attempt to evaluate any management alternatives, however.

5.4.3 Formulation, characterization and comparison of alternatives

According to the conceptual approach, economic analysis of environmental problems usually requires the evaluation of some action or policy to determine who would be affected, how they would be affected, and to what extent. Therefore, it includes the steps in which alternatives are formulated (Section 3.3.4), analyzed and characterized (Section 3.3.6) and then compared to one another (Section 3.3.7). In the Clinch Valley case study, the economic analysis had to examine management alternatives, even though the ERA had not done so. The economic analysis specified two hypothetical agricultural policies (in addition to a status quo alternative) for use in choice model construction. The apparent coherency of the choice model results suggests that respondents understood the proposed policies and choice sets and that the model is valid. However, it should be understood that the model *per se* does not characterize a specific alternative. Rather, it is a flexible, albeit semiquantitative, tool that could be useful for *comparison* of specific policies *after* they had been analyzed and characterized, as Figure 5-8 illustrates.

Figure 5-8 compares the analytic processes used in two steps, analysis and characterization of alternatives and comparison of alternatives, with those of a hypothetical example that was presented in Figure 3-3. In the hypothetical example, the ecological risks, economic effects, and health or other (sociocultural) effects of the management alternatives were analyzed quantitatively to the extent feasible. Endpoint changes that could not be quantified were expressed qualitatively. A stated preference study was used to value the nonmarket welfare effects of the alternatives and improve the estimation of their net social benefits (Section 3.4.2).

Methods used in this case study comprise a subset of those described in the example. Although the Clinch Valley W-ERA quantified relationships between land uses and ecological endpoints, the endpoint changes expected to result from the two riparian management policies introduced in the economic study were not quantified. Similarly, the financial costs and other economic effects of implementing the policies were not analyzed. Equity issues were not examined, and human health or other effects were not considered relevant to this case study. The stated preference survey used qualitative language to describe expected ecological improvements, whereas both the cost attribute and the attribute describing potential regional impacts on agriculture were numerical (Table 5-4 and Appendix 5-A).

As a result, the choice model derived from the stated preference study would be capable of comparing the benefits of these or other policies only after additional work was done. The analysis and characterization of real alternatives would require the following additional steps:

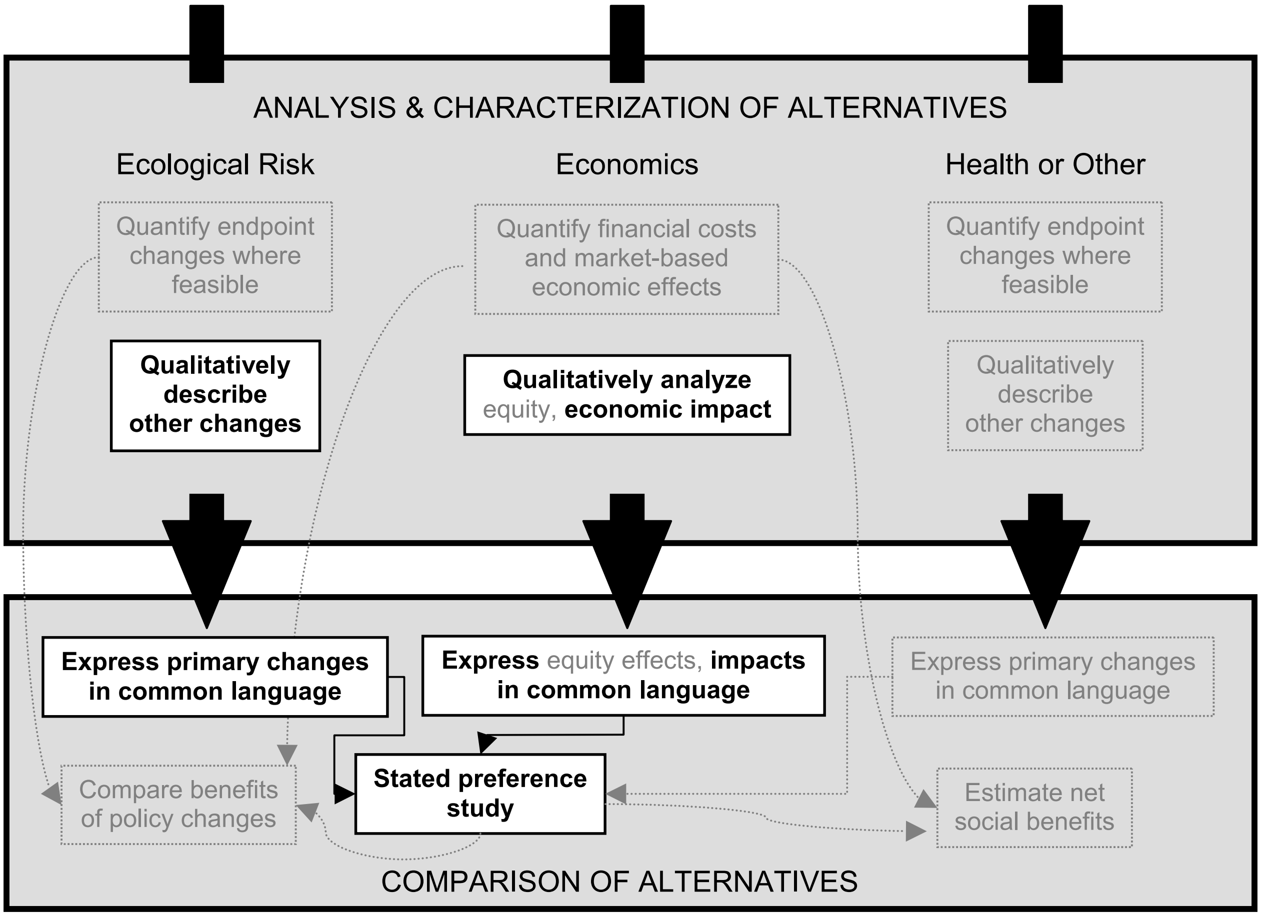

FIGURE 5-8

Techniques used for analysis, characterization, and comparison of management alternatives in the Clinch Valley watershed, as compared to the example shown in Figure 3-3. White boxes and bold type show features included in this analysis.

- determination of the decision context, including who could make the decision to implement a given alternative, how they would decide, and who would stand to gain or lose as a result (as part of planning, see Section 3.3.1)
- detailed formulation of the alternatives, including design of structural (e.g., fencing) and nonstructural (e.g., institutional) implementation measures (see Section 3.3.4) determination of the ecological outcomes (efficacy, in terms of instream biological response), economic outcomes (costs, including opportunity costs) and uncertainties of the policy (see Section 3.3.6).

Using the choice model as a comparison tool would present several additional challenges. Since the actual efficacy of a given exclusion zone for enhancing aquatic life can be estimated only with substantial uncertainty, it would be difficult to determine how a given, best estimate of increase in IBI should be evaluated in the choice model if the available choices are "partial" and "full" recovery. Respondents ascribed statistically significant value only to "full" recovery. Yet even a substantial, predicted increase in IBI would not necessarily signal a recovery of extirpated species (and certainly not of extinct species), and implementation of an exclusion zone would not reduce the very substantial risks from, e.g., transportation spills; therefore it would be hard to rate *any* agricultural policy as leading to "full" recovery. Similar problems would be encountered in coding the effects of an actual policy on sportfish and songbirds. Ultimately there would be heavy reliance on expert judgment to interpret the ecological data and to apply the choice model.

Nonetheless, the apparently successful development of this choice model suggests that models of this type can be used for comparative welfare analysis of watershed management policies. What remains unanswered, however, is the important question of whether welfare estimates are useful to decision-makers in a given case. Whereas large water resource development projects may require welfare estimates, other kinds of decisions may not. For example, if biodiversity protection in the upper Clinch Valley will continue to depend largely on success by organizations such as The Nature Conservancy at acquiring federal grants for voluntary riparian protection programs, and private funds for land acquisition, as is presently the case, it is not clear that welfare estimates are needed. For any other protection mechanism under consideration, the decision context specific to that mechanism would need to be examined to determine what information is needed for decision support.

5.4.4 Adaptive implementation

The conceptual approach for integration suggests that when uncertainties are great, management decisions should be implemented in an adaptive fashion, with continual reevaluation of effectiveness and, as necessary, redesign (Section 3.3.9). The nature and magnitude of biological response that may result from any program of riparian zone protection are uncertain. However, programs can be adaptively designed in such a way that early stages of implementation will yield the information needed to resolve specific questions and improve the effectiveness of later stages. Riparian dimensional analysis indicated that the instream impacts of riparian land use were most observable over a downstream distance of 500-1500 m (see Section 5.2.3.2). This suggests that stream reaches of appropriate lengths in different subdrainages could be pre-selected as treated and untreated replicates, with protection efforts targeted accordingly. Such an approach could yield valuable information on the amount of investment required to meet voluntary or regulatory goals for stream quality improvement in the upper Clinch Valley and other, similar watersheds.

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APPENDIX 5-A

Excerpt from Survey Administered by the University of Tennessee: Explanation of Hypothetical Agricultural Policies and their Potential Impacts

Background Information on the Clinch River Valley

The upper Clinch and Powell Rivers represent some of the last free-flowing river segments in the Tennessee River system. Together, they drain approximately 3800 square miles of land area. The Clinch and Powell Valley has one of the most diverse concentrations of freshwater mussels and fish species of any river in North America. Many of the valley's mussel and fish species are on the decline. Twenty-two mussels and eleven fish species are listed as endangered or threatened. Moreover, the Clinch River Valley has many species that are found nowhere else. Of the 50 mussel species that are listed by the U.S. Fish and Wildlife Service as "Threatened" or "Endangered", 16 are found in the Clinch River Valley.

Ecologists believe that biodiversity is important for a number of reasons, including its contribution to the health of the ecosystem (diverse ecosystems can better withstand and recover from stressors such as drought). Mussel species are good indicators of the health of the ecosystem. Because mussels are very sensitive to pollution, poor water quality will often affect mussels before it has an impact on other species in the river and before it has a direct impact on human health.

Although employment in the region is increasingly migrating to the manufacturing, service, and tourism sectors, the economy of the valley has historically been based on coal mining and agriculture. More than 40% of coal production in Virginia occurs within the Clinch/Powell Valley and much of the discharge of pollutants in the region is not regulated.

The combined effects of raising livestock, pesticide runoff and soil erosion from farming, forest clearing for development, coal mining and processing, discharge from sewage treatment facilities and septic tanks, chemical spills, runoff from roads, parking lots, and chemically treated lawns decrease water quality and reduce mussel and fish abundance and diversity.

Evaluating Changes in Agriculture to Protect the Environment

One cause of reduced water quality in the river is that livestock get into the river, crushing mussels, eroding river banks, and muddying the water. Intensive cultivation of crops near the river allows fertilizers, pesticides, soil and other substances to contaminate the river as well.

These problems could be lessened by the development of an "agricultural free zone" in the immediate proximity of the river. This zone, where crop planting and grazing would be restricted, could be of different widths. In our study, we ask you to compare the present case of no agriculture free zone with two alternative zone sizes: a zone 10 yards wide on the Clinch and 5 yards wide on tributaries or a zone that is 25 yards wide on the Clinch and 10 yards wide on tributaries.

Farmers who keep cattle would need to construct fences to keep the livestock out of the exclusion zones. Fences would keep the cattle from trampling the mussels, reduce erosion and sedimentation of the river. Trees would shade the river water, reducing its summertime temperature and increasing the dissolved oxygen level, which would benefit aquatic life. As the pastures revert to more naturally occurring types of vegetation, songbird and wildlife populations could increase. The construction of fences and substitute watering facilities for the cattle, and the loss of the use of the land are costly for farmers. Farmers who grow crops would not be able to plant in the zones, which may be among their most fertile (and flattest) land holdings.

However, the farmers need not bear the full cost of the policy. A pilot project has been underway where non-profit organizations such as The Nature Conservancy have been compensating farmers who construct fences and take lands near the river out of production. This type of project could be expanded and funded through a small increase in taxes for everyone in the Clinch Valley. The questions below ask respondents to compare possible alternative policies. One primary difference among the policies is the extent to which the farmers or the taxpayers bear the costs. Farmers could be fully or partially compensated for their losses. Another set of differences involve the levels of the environmental characteristics. These changes in agricultural practices may have effects on aquatic life, sportfish, and songbirds. The ranges of these effects that we would like you to consider are as follows:

Aquatic life: includes all non-game fish and mussels. Changes are in terms of diversity, abundance and distribution throughout the watershed.

Continued Decline = continued decreases in diversity, abundance and distribution in the Clinch River and its tributaries.
Partial Recovery = some improvement in the Clinch River, but no improvement in tributaries
Full Recovery = improvement in the Clinch River and its tributaries

Sportfish: Includes smallmouth bass, trout, etc. Changes are in terms of number and average size.

No change = current numbers and distribution of sizes
Increase = 20% increase in Clinch and tributaries
Decrease = 20% decrease in Clinch and tributaries

Songbirds: Changes are in terms of variety of species and number of birds found in the Clinch River Valley

No change = current numbers of birds and varieties of species in the valley
Increase = 20% increase in numbers of birds in the valley

Agricultural income: Changes are in terms of lost income in the agricultural sector of the Clinch River Valley economy. These losses would accrue to farmers in the 21 counties that are part of the valley as a result of decreased production.

No change = no change in agricultural income
Small decrease = $1 million/year total decrease in production. This represents less than 1 percent of total farm income for the valley.

Cost to household: One way of financing improvements to the quality of the Clinch River is to ask residents of the valley to share in the costs of protection. If you live in the Virginia portion of the valley, this could be implemented through small changes in state income taxes. If you live in the *Tennessee portion of the valley*, this protection could be paid for through small *changes in local property taxes*.

APPENDIX 5-B

RANDOM UTILITY MODEL

Using random utility theory, one can model discrete choices assuming that individuals make choices that maximize their utility, or well-being.[1] If the utility of alternative *i* is greater than the utility of alternative *j*, the individual will choose *i*. Utility is composed of both deterministic (environmental quality, income, etc.) components and random, individual-specific, components that are unobservable to the researcher. The random utility model (RUM) framework is directly estimable from conjoint rankings and choice models.

Following Roe et al.[2] and Stevens et al.[3], the utility of a management program *i* is given by

$$\mathrm{U}^i(q^i, z) \qquad \text{5-B-1}$$

where the utility (U) of program i for the individual is a function of the attributes (*q*) of i and where *z* represents individual characteristics. While utility is an interesting measure of preferences, it is not particularly valuable because it does not reflect the trade-offs, financial or otherwise, that individuals must make in order to consume a bundle of goods. Thus one typically considers the indirect utility function, which expresses utility as a function of income and prices:

$$\mathrm{U}^i = v^i(p^i, q^i, m, z) + \varepsilon^i \qquad \text{5-B-2}$$

where *v* is indirect utility and *p* and *m* represent price of the state of the world *i* and income of the individual, respectively.

Then the standard RUM can be estimated from the discrete choice conjoint data using conditional logit:

$$\Pr(i) = \Pr\{v^i(p^i, q^i, m, z) + \varepsilon^i > v^0(p^0, q^0, m, z) + \varepsilon^0\} \qquad \text{5-B-3}$$

The probability that the program having attributes *i* is chosen is the probability that the indirect utility of program *i* plus a random, unobservable error is greater than the indirect utility of program *0* and its error term.

Then *v* is estimated using a linear functional form of the indirect utility function, by means of the conditional logit model specified generally as:

$$v = const + \beta_1 Attributes + \beta_2 Socioeconomic \qquad \text{5-B-4}$$

The stylized model in Equation 5-B-4 generates the probability of choosing a particular option given the levels of attributes of the option and the individual's (socioeconomic) characteristics. The β's generated from the above equation are the coefficients associated with each of the attributes in the choice model.

To estimate the welfare impacts, or willingness to pay, for a change from the status quo state of the world to the chosen state one calculates:

$$v^i(p^i, q^i, m - CV, z) + \varepsilon^i = v^0(p^0, q^0, m, z) + \varepsilon^0 \qquad \text{5-B-5}$$

where CV (compensating variation) is the income adjustment necessary to leave the individual as well off with bundle *i* as they were with bundle *0*. Additionally, the β's from Equation 5-B-4 can be used to calculate implicit prices, or part-worths, for each variable with respect to all of the other variables in the model (see Section 5.3.3.2).[a]

[a] This is the marginal rate of substitution concept in economics upon which indifference curves are based. Simply, it gives the trade-offs that an individual is willing to make between bundles of goods while holding utility constant.

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6. SEEKING SOLUTIONS FOR AN INTERSTATE CONFLICT OVER WATER AND ENDANGERED SPECIES: PLATTE RIVER WATERSHED

6.1 WATERSHED DESCRIPTION

6.1.1 Watershed resources and impacts of development

The central Platte River floodplain in Nebraska, which includes the 130 km of river known as the "Big Bend Reach," is rich in biodiversity and ecologically complex. The reach extends from near Lexington, NE on the west to immediately below Grand Island on the east. Nested within the Platte River watershed (Figure 6-1), which encompasses 223,000 km^2 (86,000 mi^2) in Colorado, Wyoming and Nebraska, the central floodplain occupies 13,280 km^2 (5130 mi^2) and hosts a diverse assemblage of ecosystems, plants and animals. Approximately 50 species of mammals and several hundred species of terrestrial birds use the cottonwood-willow forests and wet meadow grasslands near the river for breeding or stopover habitat during migration.[1] Nearly one-half million sandhill cranes (*Grus canadensis*) and several million ducks and geese use the Platte River during their annual migration.[2] In addition, the central Platte River floodplain supports nine species of plants and animals that are listed as threatened or endangered, including the interior least tern (*Sterna antillarum athalassos*), the piping plover (*Charadrius melodus*) and the whooping crane (*Grus americana*), and another 12 species that are candidates for federal listing.[3] The high levels of biodiversity found in this reach are at risk, however, due to the cascading effects of reduced water flows and development on ecosystem structure and function.

Irrigation water from the Platte River and adjacent aquifers has made the Platte Valley a highly productive agricultural region, providing irrigation water to over one million acres. Water storage reservoirs such as Lake McConaughy and Johnson Reservoir have provided increased

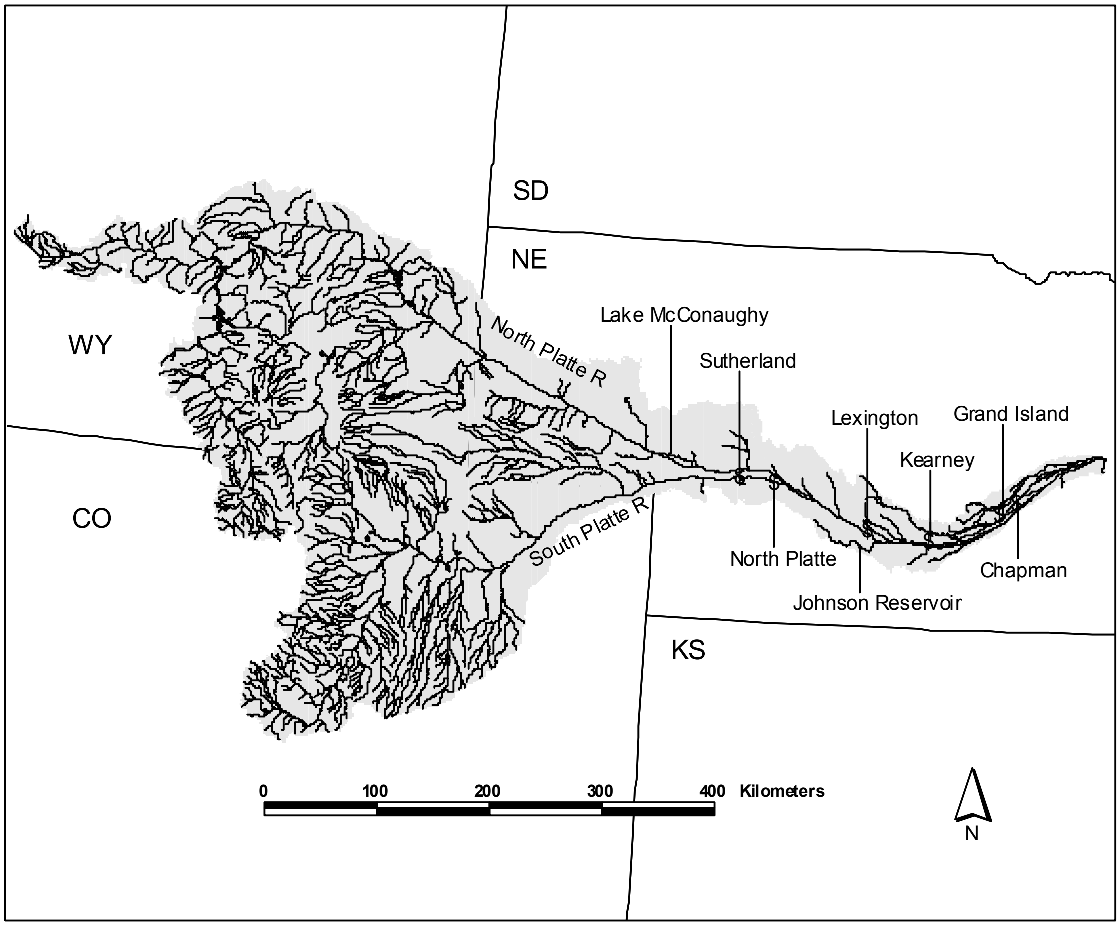

FIGURE 6-1

The watershed of the North Platte, South Platte and Big Bend Reach of the Platte River in the Great Plains of the USA. Towns and reservoirs mentioned in the text are indicated.

recreational and sportfishing opportunities, contributing to the more than two million recreational visitor days per year provided by the river. Platte River hydropower stations help meet regional energy demand by supplying 300 MW of hydroelectric power. As a result, the natural hydrologic regime has been influenced by more than 200 upstream diversions as well as by 15 dams and reservoirs on the North and South Platte Rivers, all but one of which are in Colorado and Wyoming.[4] This elaborate network of dams, diversions, and irrigation canals has resulted in a 70% decline in peak discharge.[5]

From a hydrogeomorphological perspective, the Platte River is braided stream whereby the main channel contains a network of smaller channels separated by small islands called braid bars. Braided rivers are also characterized by highly erodible banks and an abundance of sediment. In a braided system that is unregulated, the number and location of the channels and braid bars may change quickly as a function of stream discharge and sediment load. In turn, the dynamic nature of braided rivers creates a mosaic of habitats such as shifting sandbars, side-arm channels, backwaters, and temporally inundated floodplains. Combined, this rich array of habitats supports high levels of floral and faunal biodiversity. Critically, however, flood-pulsed hydrology[6] is needed to sustain this diversity of habitats and species. These flood pulses typically occur in spring as a function of snow melting in the stream's headwaters with river disturbance scouring established habitats and creating new ones. The flood pulse also maintains an important seasonal connection of the river channel to the floodplain, which distributes energy and nutrients between the river and the land, and supports ecosystem functions such as production, decomposition, and consumption.[6-8] On the Platte and other rivers, these water fluctuations also drive patterns of vegetation succession.[9-11]

In contrast to unregulated river systems, damming and other alterations to the natural flow regime alters the nature of the pulse transmitted to the Platte floodplain. As a result, the Platte has experienced reduced channel movement and environmental heterogeneity. In addition, in regulated Rivers such as the Platte, sediments become trapped behind dams, so downcutting and erosion occur in the downstream channel, further isolating the channel from the floodplain.[12-14]

Channel width on the Platte has been reduced 85-90% over the last century or so.[11,15] Establishment of *Populus* dominated forests has followed narrowing of the main channel and stabilization of river braids. Approximately half of the active channel present in the middle Platte in the 1930s had succeeded to woodlands by the 1960s due to the combined effects of irrigation, streamflow regulation and drought.[11] In total, some 9500 ha of *Populus* woodland are established in the Big Bend reach.

The significant alteration of the natural flow regime notwithstanding, high levels of faunal biodiversity are associated with the present channel structure. Two species of particular concern are Platte River populations of the least tern and piping plover -- listed as endangered and threatened, respectively by the U.S. Fish and Wildlife Service. Terns and plovers nest on large, high-elevation, barren sandbars. Historically, spring flooding during ice pack breakup would scour vegetation off of midstream sandbars, leaving the necessary open nesting substrate. Establishment of riparian forest has significantly reduced available habitat. Sandhill cranes, perhaps the flagship species of the Platte, are also highly dependent upon open channel habitat. Approximately 80% of the continental population of cranes spend about six weeks in spring staging on the central Platte River. Sandhill cranes roost in open channels and forage for invertebrates in nearby wet meadows and for waste corn in nearby farm fields.[16,17] Much has been written about the preferences of roosting cranes for open channel habitat. In general, cranes prefer roosting in shallow water and with channel widths of 500 feet or more and rarely inhabit those that are less than 150 feet. They may roost in concentrations of 20,000 per mile. Roosting on the river protects them from their predators. The issue is complex, however, because many factors are involved in the selection of roosting sites including availability of and distance to off-channel food (wet meadows and corn fields), weather, water depth, stream flow, and distance of roosts to tall vegetation.[2,18-23] Crane use has declined in the upper Platte River coincident with dramatic channel narrowing between 1930 and 1957, and has since increased farther downstream where channels have narrowed less.[24] However, large populations of cranes roost in the relatively narrow channels of the North Plate River or roost away from the river in wet meadows.[22] The effects, if any, of such displacement are unknown.

The channels are also important to a wider variety of migratory water-birds including whooping cranes and a variety of ducks and geese.[16,17,25-27] Waterfowl population estimates during migration range from 5 to 9 million individuals in spring.[28,29] Most of the migration population consists of snow geese, Canada geese, greater white-fronted geese, mallard and northern pintail.

Wet meadows that flank the Platte River support a rich assemblage of migratory and breeding grassland birds.[30] Of principal concern to this avian community are the effects of lower water tables on habitat structure and forage and particularly habitat fragmentation.[30,31] An important conservation objective is the maintenance of sufficiently large habitat patches for core-grassland (no-edge) species including upland sandpiper, bobolink, grasshopper sparrow, dickscissel and meadowlark.[30]

While alteration of the natural hydrological regime poses significant risk to many species, the establishment and evolution of the riparian *Populus* forest has created significant ecological opportunity for other species, principally those which use riparian forests. For example, based on a two year study of 72 woodland patches, Colt[32] showed that these forests support some 50 species of breeding birds, including 32 neotropical migrant species, a guild of birds that includes several species with populations at risk. Further, Colt and Jelinski (unpubl data) have preliminary findings that suggest nest success is high, and that some species are not rendered as vulnerable to deleterious edge effects (e.g., predation and nest site parasitism) found elsewhere on the Great Plains. The resulting increase in avian biodiversity as a result of altered flows broadens the number of stakeholders to include those concerned about off-channel species.

The seeming bonanza of forest bird species may substantially change, however. In less than a century, and barring a catastrophic major disturbance, the *Populus* dominated forests will almost be completely replaced via succession by equilibrium forests dominated by *Fraxinus* (ash) as Johnson[10] has predicted for the Missouri River floodplain forests. A profound biodiversity decline may result because a large proportion of flora and fauna is restricted to, or strongly associated with, *Populus* communities (Jelinski and Colt, unpubl paper). It is well established that maximum diversity of trees, birds, and small mammals occurs in older *Populus* forests midway along the sere.[9,33,34]

In summary, the flood-pulse system[6,8] that is characteristic of the central Platte River floodplain links hydrology with biological communities and ecosystem processes in complex ways.[9,11,31] Alteration of the natural flow regime for hydropower, food production, and recreation has changed the dynamic nature of the river and places some species and habitats at risk. At the same time, hydrological alteration of the Platte has created ecological windows of opportunity for a number of other species.

The effect of altered flows, habitat fragmentation, and agrochemical runoff on riparian vegetation in the central Platte River floodplain have been extensively studied,[35] whereas effects on some avian communities have barely been investigated,[2] and science is only in the early stages of predicting impacts on fish and other wildlife communities.[31,36]

6.1.2 Watershed management efforts

A long history of efforts to protect the resources of the central Platte River floodplain forms the backdrop for the ecological and economic analyses discussed in this chapter. Conservation organizations and governmental agencies have worked to improve avian habitat along the Big Bend Reach, while Federal and State agencies and various stakeholders have sought ways to resolve enmeshed conflicts between economic demands for water withdrawal and environmental needs for increased, and seasonally varying, instream flows as determined under the Endangered Species Act (ESA). Over the past 25 years, a number of management initiatives, often backed by technical analyses, have been tried.

To improve habitat suitability for cranes, waterfowl and native grassland birds, the National Audubon Society, Platte River Whooping Crane Maintenance Trust, and The Nature Conservancy have acquired tracts of wet meadow and river channel. They have eliminated roads, fences and buildings and have consolidated land units to reduce disturbance and habitat fragmentation. The Natural Resource Conservation Service of the U.S. Department of Agriculture (NRCS) and the U.S. Department of Interior's Fish and Wildlife Service (USFWS) have cooperated to restore wet meadow and open-channel roost habitat for cranes by removing woody vegetation from sandbars in the river channel. These actions have not been without controversy, however, as the mechanical removal of some tracts of late seral vegetation to recreate early-successional habitats has favored the requirements of certain wildlife species while destroying established habitat for others. There is also scientific disagreement over the extent to which riparian land management can effectively substitute over the long term for restoration of stream flow. [23,37]

Concerning required flows, some scientists contend that high stream flows are needed periodically to prevent vegetative growth on sandbars and sustain the wide and shallow riverine habitat preferred by whooping and sandhill cranes,[38,39] whereas others contend that such scouring flows are of little value and may actually be harmful in the case of fish, because scouring flows lead to lower reservoir levels and higher water temperatures.[40] The terms of the legal debate over stream flow are defined by ESA provisions that prohibit any Federal action jeopardizing the continued existence of a species designated as threatened or endangered, and provide that USFWS determine species' requirements based on the best available scientific information. The USFWS has determined that an additional 417,000 acre-feet (514 hm^3) per year of water is needed to meet endangered species needs for the Big Bend Reach in a wet-to-average year.[a,3] Absent any agreement as to how to make up that deficit, this determination is sufficient to preclude any major water consuming action that constitutes a federal nexus. In other words, the U.S. Forest Service (USFS) water leases in Colorado cannot be easily renewed; Wyoming cannot pursue additional, federally permitted upstream water storage projects that would increase consumptive use; and the public power districts in Nebraska cannot be assured of getting a long-

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[a] This annual volume does not include less frequent flow recommendations such as a 5-year peak flow of 16,000 cfs for channel maintenance.

term hydropower license from Federal Energy Regulatory Commission (FERC) unless some accommodation of the competing demands can be made.

Stakeholder groups have been actively involved in management discussions that have occurred in the context of water right litigation, power plant licensing hearings, legislative debates and other venues (FERC, 1998). Environmental interests in all political jurisdictions (Colorado, Wyoming, Nebraska and the USFWS) tend to agree on the need for increased and re-regulated stream flow and management of riparian lands for endangered species protection. Irrigation interests are much more parochial both between and within states. Upstream surface water irrigators have sought the right to continue irrigating and, in some instances, the right to develop additional acreage. Downstream surface water irrigators want their water supply protected against additional depletions from upstream irrigation or environmental demands. Groundwater irrigators in all locations have sought the right to pump at will, irrespective of stream flow considerations. Hydropower interests want high reservoirs to maximize feet of head and would like to make reservoir releases during the summer months when electricity is worth the most. Coal fired electric utilities want assured cooling water supplies and expansion opportunities. Finally, recreation interests have mixed demands, including moderate reservoir storage levels, stream flows that sustain fishing and waterfowl hunting, and easy access to the river and to bird watching opportunities.

Since 1976 the Nebraska Department of Water Resources (DWR) has held over 400 days of public hearings to address proposed diversions or requested instream uses of Platte River water. From 1983-1997, the public power districts in Nebraska were in negotiations with the FERC over the relicensing of Lake McConaughy. In addition, from 1986 - 2001 the states of Wyoming and Nebraska were in litigation over the interstate allocation of Platte River water.

The struggle to manage the Platte system has led to several attempts to facilitate resource management decisions, including empirical modeling with and without stakeholder input, several negotiation formats, multi-state litigation and, most recently, a tristate-federal Cooperative Agreement[41] that takes an interim, adaptive management approach to the problem. One of the first organized attempts to reach a compromise solution was an adaptive environmental assessment process which began in 1983. Called the Platte River Forum, this approach involved identifying a group of experts and stakeholders and assembling them in a single location for one week. This group first identified the relevant impact variables and policy options. Then, with the help of experts, the associated technical relationships were described in mathematical terms and computerized. The idea was that stakeholder participation and input would lead to a widely supported simulation model and agreement regarding the consequences of management options.[42] This expectation proved to be invalid. Not only did participants fail to agree on all the facts, but even when there was general agreement on how the natural system worked, differing value judgments and varying objectives prevented completing a model that was very useful for determining how the water should be used.[43]

The Platte River Forum was responsible in part for the formation of a small research group to develop a multi-objective model of the Platte. This model was built by a group of university professors without stakeholder involvement.[44] Whereas the Platte River Forum focused on the physical aspects of the river system and considered only a small set of alternatives, the multi-objective model focused on the delineation of trade-off curves for numerous alternatives. The intent was to improve on the Platte River Forum by producing additional information for decision-making and to do so without the inefficiencies and biases of a committee of 30, many of whom represented stakeholder interests rather than areas of expertise.

The outcome of the multi-objective modeling approach can best be characterized as good science that was unused and ineffective. The scientists involved, operating independent of political pressure, were able to produce a credible operational model, but the results were not embraced by any interest group or decision-maker.

A third attempt to resolve the water management problem involved relicensing of hydropower plants. From 1986 until a provisional hydropower license was issued in 1997, the Central Nebraska Public Power and Irrigation District and the Nebraska Public Power District were involved in an intensive effort to get the FERC to relicense their Platte River hydropower facilities. The central issue was protection of threatened and endangered species, but NEPA requirements associated with licensing a public resource also meant that broader fish and wildlife issues, including sandhill crane habitat, had to be addressed. The major hydropower facility involved is part of the Kingsley Dam which creates Lake McConaughy.

Lake McConaughy is the largest reservoir on the Platte River and the closest one to the endangered species habitat. Historically, Lake McConaughy has been used to directly irrigate over 200,000 acres (77,000 ha) and to enhance the groundwater supply for an additional 300,000 acres (112,000 ha).[45] It has also been managed as a fishery in cooperation with the Nebraska Game and Parks Commission and is a significant recreational resource drawing over 600,000 annual visitors per year. For nearly 50 years, however, the water entering Lake McConaughy was managed in a serial dictatorship with irrigation receiving first priority for the water, followed by hydropower and recreation. Endangered species were not considered. This all changed when the original hydropower license expired in 1987. FERC required the Districts to address wildlife habitat maintenance and enhancement, which led to extensive study by the Districts and by environmental interest groups, and eventually to intensive negotiations between the Districts, environmental interests and FERC. However, the parties were unable to agree on how to balance endangered species with other needs. Licenses were nevertheless issued provisionally, with a requirement that the districts' operations be coordinated with the proposed Cooperative Agreement.

As the pressures for reallocating water to meet endangered species needs mounted, Nebraska interests sought to broaden the responsibility for meeting these needs to include Colorado and Wyoming. Of the two million acres irrigated with surface water within the Platte Basin, Colorado has 56 percent, Wyoming 12 percent and Nebraska 32 percent. It seemed unfair to Nebraska water interests that they should have to meet endangered species needs without appropriate contributions from Colorado and Wyoming.[45] At the same time Colorado was facing endangered species problems with Forest Service water rights and with potential irrigation projects, while the threat of subjecting U.S. Bureau of Reclamation projects to consultations under the ESA had eastern Wyoming and western Nebraska irrigators nervous.[45] All three states found that cooperation was in their mutual interest and negotiated the Cooperative Agreement, initiated in 1994 and signed on July 1, 1997.

The Cooperative Agreement constituted a multistate-federal effort to protect Platte River endangered species without unduly constraining the availability of water for other uses. It established a preliminary agreement to increase instream flow by an average of 130,000-150,000 acre-feet (160-185 hm^3) and to acquire an initial 10,000 acres (3,900 ha) of an eventual 29,000 acres (11,200 ha) of riparian habitat, but did not set forth where all of the water would come from nor what land would be acquired. The participants had three years to study alternatives and to agree on sources of water and land, including a distribution of the costs. (As of this writing, however, progress has been slow and the period for reaching agreement has been extended to

June, 2003 and may be extended further.) If agreement is reached, the plan is to be put in place and monitored for 10-13 years to determine how well the program is meeting endangered species needs. If an agreement is not reached, the public power districts in Nebraska may lose their provisional hydropower licenses, holders of water right leases on Forest Service lands will find renewal very difficult, new surface water development in all states will be difficult, if not impossible, and actions to protect endangered species will be further delayed.

Whether the Cooperative Agreement is successful or not remains to be seen, but thus far none of the management approaches used have led to a comprehensive resource management plan that addresses the conflicting demands of competing interest groups.

6.2 ECOLOGICAL RISK ASSESSMENT

6.2.1 Planning

Concern over threats to the valued biodiversity of the central Platte River floodplain, coupled with evidence that various agencies and stakeholders would be willing participants (Table 6-1), motivated the U.S. Environmental Protection Agency (USEPA) in 1993 to establish an interdisciplinary workgroup to begin a watershed ecological risk assessment (W-ERA). The goal was to obtain a better understanding of how the central Platte River landscape and associated flora and fauna are being impacted by water withdrawal and other stressors. The workgroup was composed of individuals with disparate interests and responsibilities and many years of experience working in the central Platte River watershed. The planning process included face-to-face dialogue between assessors and resource managers, a group tour of the watershed, symposia, public meetings, focus group meetings and teleconferences.

Recognizing that any protective management actions would have to be weighed against the need for human uses, the workgroup developed the following management goal for the

TABLE 6-1 Participants in planning for the central Platte River floodplain W-ERA
Central Nebraska Public Power and Irrigation District
Nebraska Public Power District
Nebraska Department of Environmental Quality
Nebraska Natural Resources Commission
Central Platte Natural Resources Districts
Nebraska Game and Parks Commission
Tri-Basin Natural Resources Districts
Nebraska Department of Agriculture
The Nature Conservancy
Prairie Plains Resources Institute
Platte River Whooping Crane Maintenance Trust (PRWCMT)
University of Nebraska -- Lincoln and Kearney
US Fish and Wildlife Service
US Geological Survey
US Department of Agriculture

watershed: protect, maintain and, where feasible, restore biodiversity and ecological processes in the central Platte River floodplain, to sustain and balance ecological resources with human uses. The management goal is a qualitative statement that addresses concerns expressed by various agencies and management organizations as well as the floodplain residents and other stakeholders.

6.2.2 Problem formulation

This section summarizes the problem formulation exercise conducted for the central Platte. The intricacies of that process, and the limitations of the resulting analyses presented in the following section, illustrate the difficulty of narrowing a broad management goal for a large and complex system to a tractable set of risk assessment problems.

The management goal was interpreted by representatives from USEPA's Region VII and Office of Water, the USFWS, the U.S. Geological Survey and Nebraska officials (listed in Table 6-1) into potentially implementable environmental management objectives (Table 6-2). A more detailed description of the watershed than that presented in Section 6.1 was developed, along with a description of the environmental problems in the watershed. The environmental problems emanate from a combination of physical and chemical stressors. Of the many human-caused stressors thought to be interfering with attainment of the goal, eight principal stressors were selected by the workgroup (Table 6-3), using a Delphi ranking technique[46] that documents iterative group input and helps groups reach consensus. Nine ecological assessment endpoints, representing three spatial scales, were selected (Table 6-4) that met the criteria of (a) relevance to environmental management objectives, (b) ecological relevance and (c) susceptibility to stressors (see Section 2.1.1.2).

TABLE 6-2

Eleven environmental management objectives that are implicit in and required to achieve the management goal

Affected Area		Environmental Management Objective
Channel	1	Restore and maintain stream channel dynamic equilibrium
	2	Maintain sufficient flows to prevent high temperatures detrimental to native fish populations
Riparian Forest	3	Maintain range of successional stages of forest vegetation
Backwaters	4	Maintain and reestablish backwater ecosystems
	5	Maintain and restore hydrologic connectivity between river channels through surface flows
Floodplain	6	Maintain hydrologic connectivity between river channels and wet meadow ecosystems
	7	Maintain and reestablish natural diversity in wet meadow systems
	8	Maintain and reestablish natural diversity in native upland systems
Landscape	9	Protect and where feasible reestablish the mosaic of habitats in the central Platte River floodplain to support key ecological functions and native biodiversity.
	10	Maintain diversity of water-dependent wildlife including migratory and nesting birds, mammals, amphibians, reptiles and invertebrates.
	11	Prevent toxic levels of contamination in water consistent with state water quality standards

TABLE 6-3 Principal stressors (and their primary sources) in the central Platte River floodplain
Altered surface water regime (dams and diversions)
Truncated sediment supply (dams and diversions)
Altered ground water regime (dams, diversions, groundwater withdrawal and irrigation)
Physical alteration of habitat (land conversion to agriculture, including drainage of wet meadows, and clearing of vegetation for wildlife management)
Nutrients (fertilizer use)
Toxic chemicals (agricultural biocide use)
Harvest pressure (fishing, seining, waterfowl hunting)
Direct disturbance (roads, off-road vehicles, bird watching)

TABLE 6-4

Ecological assessment endpoints for the central Platte River floodplain W-ERA.

Landscape scale	Floodplain landscape mosaic structure, function and change
Habitat scale	Open channel configuration and distribution for migratory birds
	Side channel and backwater area and connectivity to main channels
	Riparian vegetation successional stage, areal extent and dispersion
	Wet meadow composition and abundance
Organism/Population level	Sandhill crane and waterfowl diversity, abundance and dispersion
	Core grassland breeding bird diversity and abundance
	Amphibian survival and reproduction
	Riverine and backwater fish and invertebrate survival and reproduction

As in other risk assessments discussed previously, detailed conceptual models, developed for each endpoint, were used to hypothetically attribute stressors to their sources and to explain their impact on the assessment endpoints. Three of the nine assessment endpoints, or representative elements of them, subsequently were selected as priorities for detailed quantitative analysis. Those endpoints, and the corresponding risk hypotheses that were derived from the conceptual models, are presented in Table 6-5. These three were selected because they capture the predominant concerns regarding birds and unique habitat in the floodplain and because they crystallize water and riparian management conflicts. All three are linked to the fact that lower rates of flow reduce channel habitat for species such as sandhill cranes, piping plovers and least terns[2,17,18,47] and reduce shallow groundwater levels, thereby desiccating wet meadows and reducing habitat diversity.[48] However, lower flows promote the establishment of riparian forests favored by other avian species.

The embattled nature of the Platte River management problem was evident during the problem formulation process. An initial draft of the planning and problem formulation report was presented to, and amended by, the stakeholder group in February of 1996. Subsequently, the draft was further revised by the risk assessment team, in accordance with USEPA's concurrently-developing ERA guidance. Upon release of the revised draft,[49] some of the stakeholders considered the revised draft overly environmentalist in tone and a breach of group process, and they formally complained to USEPA by way of their Congressional representatives. To some extent, this disagreement reflects a divergence in values and objectives between the larger environmental community and those who live in the region. As such, it is characteristic of the problems encountered when the benefits of environmental improvements accrue to a broad community, while most of the costs are incurred locally.

TABLE 6-5

Selected assessment endpoints and stressors and the associated risk hypotheses developed during problem formulation for the central Platte River floodplain W-ERA.

Priority Assessment Endpoints	Principal Stressors	Risk Hypotheses
Riparian vegetation successional stage, areal extent and dispersion	Altered surface water regime	1. Lower flows have led to reduced reworking of channels, greater cottonwood regeneration, less heterogeneity of riparian vegetation.
	Truncated sediment supply	2. Reductions in sediment may alter development of river braids by lowering river bed elevation, decreasing sediment deposition on floodplain, increasing stability and reducing riparian heterogeneity.
	Physical alteration of habitat	3. Removal of riparian woodland vegetation by mowing and cutting reduces patch size and diversity of riparian vegetation.
	Toxic chemicals	4. Herbicide drift and runoff from agricultural fields have caused physiological stress and perhaps increased mortality in riparian vegetation.
Core grassland breeding bird diversity and abundance	Altered ground water regime	5. Lowered water table reduces diversity of wet meadow vegetation and renders adults, eggs and young more susceptible to predation.
	Physical alteration of habitat	6. Loss of habitat, reduction of patch size, and fragmentation of habitat may lead to decline of species requiring large wet meadows.
Sandhill crane abundance and distribution	Altered surface water regime	7. Lower flows lead to additional woody plant establishment, channel narrowing and deepening, and roosting habitat fragmentation. These changes reduce roost suitability, increase crowding and may increase susceptibility to disease or other catastrophic events.
	Truncated sediment supply	8. Reductions in sediment supply reduce channel braiding and thus open-channel roosting habitat.
	Physical alteration of habitat	9. Wet meadow conversion to crops has fragmented crane foraging, loafing and resting habitat; channelization has reduced roosting habitat suitability.
	Direct disturbance	10. Auto and rail traffic and crane-based tourism disturb migrating cranes.
	Altered ground water regime	11. Lowered water tables reduce the production of wetland invertebrates, tubers and seeds that provide forage for migrating cranes.

Source: Jelinski[31]

6.2.3 Analysis

Because of reassignments and shifting priorities, only a portion of the quantitative exposure and stress-response analyses that were contemplated could be completed, even for the reduced list of three assessment endpoints. This section presents those partial analyses.

6.2.3.1 Riparian vegetation successional stage, areal extent and dispersion

The risk hypotheses attributed fragmentation and loss of heterogeneity of riparian vegetation to reductions in instream flow and sediment supply, as well as to riparian habitat management measures, including mowing to create crane roosting habitat. It was also hypothesized that agricultural herbicide use may pose additional stress. Reductions in mean annual flow, peak flow and sediments in the central Platte River during the period of regulation are well documented, as are reductions of active (unvegetated) channel area, increases in wooded area and decreases in wet meadow area since the onset of regulation.[3,11,29,35,50] Therefore, the veracity of hypotheses 1 and 2 (Table 6-5) is not much questioned, but efforts to develop quantitative relationships between these variables, to enable estimates of risk, were not completed. Analysis of herbicide impacts on riparian vegetation was not undertaken, nor was there an analysis of riparian management effects on patch dimensions.

6.2.3.2 Core grassland breeding bird diversity and abundance

Risk hypotheses postulated that lowered ground water levels and habitat destruction and fragmentation reduced habitat suitability for, and survival of, several grassland nesting species. Therefore, an analysis of habitat use data was performed. Helzer and Jelinski[30] surveyed 45 and 52 grassland patches, in 1995 and 1996 respectively, in the central Platte River valley. Patch size ranged from 0.12 to 347 ha; roughly half of these meadows were used for grazing, the others for haying. In each patch, four randomly selected, 100-m transects (4 ha total area) were surveyed twice between May 17 and July 5; and species that are exclusively grassland nesters were censused. Where intended sampling area exceeded patch size, patches of similar size characteristics were combined. Patch area and perimeter were determined using aerial photographs and digital planimeter. Thirteen wet meadow breeding species were found during the two field seasons; the six most common were used in species occurrence models, and all 13 were used in species richness analysis.

Occurrences of all six common species and species richness were most strongly (and inversely) correlated to perimeter-area ratio, indicating that habitat use by wet-meadow nesting species is maximized in patches that provide the most abundant interior area, free from edge effects. These findings directly supported hypothesis 6 (Table 6-5). Since wetness or vegetational diversity within these patches was not measured, hypothesis 5 was not evaluated.

An analysis of diversity and abundance of 50 woodland breeding bird species was also carried out[32] but was not completed (Colt and Jelinski, unpublished data). During the 1995 and 1996 breeding seasons, birds were censused in 72 woodland habitat patches ranging in size from 0.02-44 ha and were analyzed in relation to five spatial variables (related to patch size and shape) and 15 habitat structural variables (e.g., tree species richness, average tree basal area, canopy height, tree density, percent area flooded). In preliminary findings (May 2000 communication by D. Jelinski to V. Serveiss and R. Fenemore), both richness models and occurrence models (the latter were significant for 24 species) tended to indicate that although structural variables (including canopy cover, shrub stem density and percent area flooded) were significant for some species, spatial variables related to patch size were more important in general. These findings suggest that a statement similar to hypothesis 6 can be made for woodland avifauna.

6.2.3.3 Sandhill crane abundance and distribution

Over a six-week period during spring migration, approximately 500,000 sandhill cranes stage in the central Platte River floodplain, with an individual staying about 2 - 4 weeks to rest and accumulate fat reserves. Cranes are known to roost in the evening in broad, shallow segments of the river channel. They prefer channels at least 150 m wide and 10-15 cm deep, with unobstructed views. Though they will roost in channels less than 150 m wide, they avoid those less than 50 m in width.[23,51] Faanes and LeValley[24] evaluated population changes among four staging areas and found that a west-to-east shift had occurred. This shift was attributed to loss of roost habitat in some of the western river segments and to scouring river flows and human removal of woody vegetation providing more desirable roost site in some eastern segments. Controversy exists, however, as to whether the river channel is now in a state of equilibrium with respect to suitability for crane roost habitat, or in a state of decline.[11,39]

Risk hypotheses attributed reductions in roost suitability to reduced river flows, reduced sediment supply, reduced acreage (and wetness) of wet meadows, channelization, and direct disturbance (Table 6-5). The Cadmus Group[52] attempted to evaluate relationships between sandhill crane distribution and habitat and to develop a model capable of predicting future changes in crane use of staging habitat in the central Platte River valley. Using habitat data determined in 1982,[29] coupled with USFWS annual, one-day crane census data for the flanking years 1980 - 1984, evaluations were performed by bridge-to-bridge river segment ($N = 15$), by river reach ($N = 10$) and by crane staging area ($N = 4$). Associations by bridge segment were weak, most likely because bridge segments are not ecologically meaningful. On the river reach scale, mean unobstructed channel width showed the best relationship to crane density ($r^2 = 0.45$; $p<0.05$), while the density of wet meadows (ha of wet meadows per river kilometer) showed a rather weak relationship to crane density. When data are aggregated by staging area, the relationships improve, and crane density is a function of both mean channel width and the density of wet meadows, in a two-step relationship. First, if mean channel width is less than about 50 m, cranes will not be present. For staging areas with mean channel widths greater than 50 m (i.e., Kearney to Chapman, Lexington to Kearney, Sutherland to North Platte), the following best-fit regression model was obtained:

$$\text{ABUND} = 318 + 3.74\ \text{MEADOW} - 1.39\ \text{ALFALFA} \qquad 6\text{-}1$$

where ABUND is crane density (numbers/km of river) and MEADOW and ALFALFA are density (ha/km of river) of wet meadows and alfalfa fields, respectively. For this model, the adjusted r^2 was 0.754 and p was 0.0002; the standard errors of the intercept, MEADOW, and ALFALFA were 147, 1.28, and 0.67, respectively. The regression of crane abundance versus density of wet meadows alone was also significant ($p = 0.0002$; adjusted r^2 of 0.665); the best fit equation for this model was:

$$\text{ABUND} = 39.9 + 5.49\ \text{MEADOW} \qquad 6\text{-}2$$

in which the standard errors for the intercept and Meadow were 69 and 1.08, respectively. These findings are generally consistent with aspects of hypotheses 7 - 9 and 11 (Table 6-5). They demonstrate that there is an apparent threshold for acceptable channel width, above which the availability of forage habitat (especially wet meadows, and to a more limited extent alfalfa) is most important. However, data on channel widths and areas of wet meadows and alfalfa fields more recent than 1982 were unavailable to test the model, limiting its confidence and reliability.[a]

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[a] The PRWCMT has collected additional data on crane use between 1998 and 2002, but as of this writing not all of it has been converted to a useable form.

Furthermore, the relationship between the primary stressors – i.e., reductions of flow, sediment and ground water level – and either the habitat variables or crane abundance could not be investigated by this approach, and thus the analysis was not directly applicable to decisions related to water management. Data on direct disturbance (hypothesis 10) were not available for analysis.

6.2.4 Risk characterization

As mentioned above, risk analyses for the central Platte River floodplain were not completed, and therefore risk characterization, or the translation of exposure and response analyses into meaningful – and, where possible, quantitative – statements about risk, could not be carried out. Nonetheless, the W-ERA served to summarize existing knowledge about risks to a set of valued ecological endpoints in the region, to focus information needs on a set of risk hypotheses and to provide new data and quantitative relationships for several of these endpoints. These findings are potentially valuable because factual disagreements underlie some of the ongoing resource management disagreements discussed in Section 6.1.2. Whereas the questions currently driving policy are specific to the water and habitat needs of federally threatened and endangered species, the ecological risk problem was formulated more broadly to examine the ecological integrity of the region as a whole. These results do not directly address the question of target flows in the Big Bend Reach, but they do speak directly to the importance of maintaining broad, active river channels and a diverse riparian landscape mosaic – i.e., one that includes wet meadow patches with large interior dimensions and forested patches of varying seral stage – as the means to protect regional biodiversity, particularly of birds.

6.3. ECONOMIC ANALYSIS

Environmental economics often approaches environmental management problems as budget-constrained, social-utility maximization problems, in which a key role for analysis is the quantification of policy-relevant costs and benefits, including those related to nonmarket goods (see Section 2.2), so that a socially optimal policy can be found. Ecological economics often takes a similar approach – while adding a sustainability or other biophysical constraint. Experience with Platte River decision-making, however, suggests that technical analysis alone does not lead to a resource management equilibrium, either optimal or suboptimal. First, information asymmetries create principal-agent problems (see Sections 2.2.1 and 2.2.5). For example, states have an incentive to overstate their political compensation costs for providing environmental water. Second, the presence of multiple objectives and stakeholder groups means that the optimal management plan is different for each stakeholder group and that a global social optimum cannot be achieved without weighting the relative importance of each. Such weights are never explicitly assigned, but instead are implied by the decisions that are taken. A resource management equilibrium is reached only when each stakeholder group believes that the cost of further negotiations or political action exceeds the value of the expected change in outcome, a condition which closely approximates the classic Nash equilibrium in economic game theory.[53]

All participants in the dispute over environmental management in the central Platte River floodplain have a strong incentive to reach a solution. Without a negotiated solution, the federal government will have greater difficulty meeting its ESA obligations; agriculturalists could face federal imposition of very high instream flow requirements; environmentalists would encounter further delays before instream flows are increased; and the states face continued uncertainty, hampering their individual water management and economic development programs and threatening higher costs if a settlement is imposed. In spite of these incentives, the parties have been unable to reach an agreement. A case in point is the need to follow-up the general agreement reached in the Cooperative Agreement – i.e., to increase Platte River flows by 130,000-150,000 acre-feet for 10 years and to monitor the results – with a specific agreement as to how the state and Federal parties will provide and pay for the water. All stakeholder groups continue to argue over technical issues and to take strategic positions designed to improve the resource management outcome from their point of view.

Recent developments suggest that selected game theory techniques (see Section 2.2.5) may be useful in resolving this conflict. Game theory occasionally has been applied to water resource management problems during the last decade. Becker and Easter[54] used game theory to analyze the dependency among eight states and two provinces concerning water diversions from the Great Lakes. Diversion decisions were modeled under different scenarios with different restrictions on the lakes where diversions could occur. The results suggested that states do not necessarily divert water because they stand to gain relative to the status quo, but because they may lose more if they follow an alternative future strategy. In a case similar to the central Platte, Adams et al.[55] proposed game theoretic models in the form of computer simulations to investigate the likely outcome of negotiations among agricultural water users, environmental groups and municipal water users in California. Their results indicate that the outcome of the negotiation process depends crucially on the institutional structure of the game, the input each group has in the decision-making process, the coalitions of groups that can implement proposals, the scope of negotiations and the outcome if parties fail to reach agreement.

The principal appeal of game theory to the central Platte bargaining problem is that it offers the potential of inverting the problem from a case where stakeholder representatives propose solutions to each other to one where stakeholders respond to solutions suggested by game models. This increases the possibility that an equilibrium solution will be found, because all bargaining strategies are simultaneously considered and because mathematical manipulation is likely to reveal solutions that may not emerge in a round table bargaining process. Although a realistic game model for this situation is unlikely to have a solution that meets all constraints, and it will certainly not have a unique solution, the game theory approach may still have considerable merit. It forces the participants to consider the role of incentives and strategic behavior in bargaining and, if nothing else, increases the likelihood that individual stakeholder groups will pursue policy options that are attractive enough to all participants to have a reasonable potential for successful implementation.

The decision to focus the economic analysis on the Cooperative Agreement process, and to use game theory, was made by the economic research team of the University of Nebraska - Lincoln (UN-L) in their application for a USEPA grant. Some team members had a longstanding involvement with the instream-flow negotiations. After the grant was awarded, and prior to the start of work, an informational meeting was held in 1999 involving USEPA, the UN-L research team, a representative of the Nebraska Natural Resources Commission (familiar with stakeholder concerns and the Cooperative Agreement), the Platte Watershed Program Coordinator of the UN-L Cooperative Extension Service (familiar with habitat management efforts), and the lead researcher for the W-ERA. Participants were informed regarding the status of the W-ERA, the status of the Cooperative Agreement, and the proposed economic research approach.

For this analysis the central Platte management problem was defined in terms of two game models: Model I, which addresses who should provide and pay for environmental water (i.e., water reallocated to instream flow for purposes of maintaining or enhancing biodiversity), and Model II, which addresses how much water should be so reallocated. Data for Model I were obtained from available reports, whereas Model II required a survey of households in Colorado, Nebraska and Wyoming. The next sections present the methods and results of each model in turn.

6.3.1 Model I: Determining who should provide and pay for environmental water

The parties to the Cooperative Agreement (initiated in 1994 and signed in 1997 by Colorado, Wyoming and Nebraska) have agreed as to an incremental amount of instream water (i.e., 140,000 acre-feet) that would constitute a first step in the adaptive implementation of measures to protect threatened and endangered species in the central Platte River floodplain. However, they have not been able to fully agree on the source of the water or who would pay for it (as well as a number of other administrative details). This study hypothesized that an auction approach capable of addressing information asymmetries would lead to an agreement in circumstances where other negotiating strategies may break down. After examining auction techniques (see Klemperer[56] for a comprehensive review), the approach selected was a second-price, sealed-bid sequential procurement auction with descending bidding and predetermined cost shares. In a sequential procurement auction, one unit (in this case, a given quantity of water) is auctioned at a time, and a single buyer receives bids from several sellers. In a descending-bid (or English) procurement auction, price falls incrementally until only one seller remains. If the auction is of the second-price (or Vickrey) variety, the winning seller receives the second-lowest bid, which eliminates the incentive for a seller to bid higher than his minimum price. Most of the auction literature deals with auctions where a single unit is sold at a time. Sequential versions of each standard auction type exist, although their use is not well researched.[56]

The only players in this game are the three states. Environmental or agricultural interest groups are not players because their primary concern is assumed to be the amount of reallocation, not who pays and at what price. The federal government's only role is to commit to a given cost share at the beginning of the game. The predetermined cost shares define how much each state and the U.S. Department of Interior (DOI) contribute to the cash pool for purchasing environmental water. The state with the winning bid incurs an obligation to supply the water in return for a payment from the cash pool. Although the use of predetermined cost shares may be unusual or unexpected, it is consistent with the terms of the Cooperative Agreement mentioned in Section 6.1.2.

It is well-known that for a sealed-bid, second-price auction it is a dominant strategy for each player to announce costs truthfully.[56] The descending English auction design does not necessarily result in truthful revelation of all costs, but it does result in a dominant strategy equilibrium that minimizes welfare costs. All players bid until only the two lowest cost players remain; then the agent with the second lowest cost stops at his cost and the lowest cost player wins the auction with a bid equal to (or slightly below) the second lowest cost. Mathematical details and proof that the strategies result in a Nash equilibrium have been reported elsewhere.[57]

6.3.1.1 Data sources

The data needs for this model consisted of acquisition costs, third party costs and political compensation costs. Acquisition costs represent what each state will need to spend to acquire the water for reallocation to environmental uses, such as for acquiring water rights, for providing additional storage, or other costs depending on the water source. Acquisition costs were compiled from a recent report[58] prepared for use by the states and the DOI in resolving the central Platte management problem. Third party costs were assumed to be 10 percent of acquisition costs based on historical levels of unemployment and underemployment and on regional input-output model results for the central Platte region[59] and the states of Nebraska, Colorado and Wyoming.

Political compensation costs are the payments above expected opportunity costs (i.e., foregone economic benefits) that the states may demand as compensation for the political turmoil and economic uncertainties associated with agreeing to supply a given quantity of water. These values can be inferred from game results if the game is actually played, rather than simulated as in this study. For purposes of this analysis, three different levels of political compensation were defined which, based on the investigators' observation of the Cooperative Agreement Governance Committee's discussions on this issue, were expected to bound the problem: no compensation, moderate and high. Political compensation for the moderate case, expressed as a multiple of the real cost, started at near zero for the first blocks of water supplied by a single state and increased exponentially to 20 percent of real cost at 50,000 acre-feet and to 57 percent at 140,000 acre-feet of water supplied. Corresponding points on the political compensation function for the high compensation case were 40 percent of real costs at 50,000 acre feet and 113 percent at 140,000 acre feet.

Simulations assumed a cost-share policy consisting of Colorado 0.2, Nebraska 0.2, Wyoming 0.1 and the DOI 0.5. These shares are based on the initial cost allocations that were incorporated in the 1997 Cooperative Agreement between the states and the DOI. Water was procured in blocks of 10,000 acre-feet with minimum bid increments of \$0.50 per acre-foot. Results were computed for water supply quantities ranging from 10,000 to 420,000 acre-feet per year (i.e., the total increment recommended by USFWS), but all welfare comparisons were calculated for a quantity of 140,000 acre-feet, the target quantity adopted under the Cooperative Agreement.

6.3.1.2 Model I results

Water supply costs under three different political compensation policies are depicted in Figures 6-2a to 6-2c. In Figure 6-2a, the observed difference between marginal cost and bid price is the second-price gain, whereas in Figures 6-2b and 6-2c that difference includes political compensation costs as well. Under a no-political-compensation policy (Figure 6-2a) the costs are lowest, but Nebraska would need to supply 110,000 out of 140,000 acre-feet, or 79 percent of the water. This finding reflects the fact that most of the low-cost water is in Nebraska,[60] but results of preliminary multi-state negotiations to develop a water supply plan suggest that a cost minimization approach is not likely to be politically acceptable. Under these circumstances one would expect Nebraska to bid high in order to either get adequate political compensation or induce another player to supply the water, whichever comes first. Under the simulated effect of political compensation (Figures 6-2b and 6-2c), exponential increases in Nebraska's bid price, and a corresponding increase in cumulative budget costs, provide incentives that cause supply by Wyoming and Colorado to increase. However, net welfare costs (Table 6-6) increase less than budget costs, because the budget increase is largely in the form of political compensation transfers among the parties. The second price effect increases as political compensation increases, with second price gains going to those who supply the water. Most of the increased welfare costs accrue to the federal share, because they supply no water and therefore receive no second price gains or political compensation transfers.

In summary, these findings present a scenario in which a mutual supply agreement, unachievable up to this point, could be reached for a modest increase in total welfare costs (when

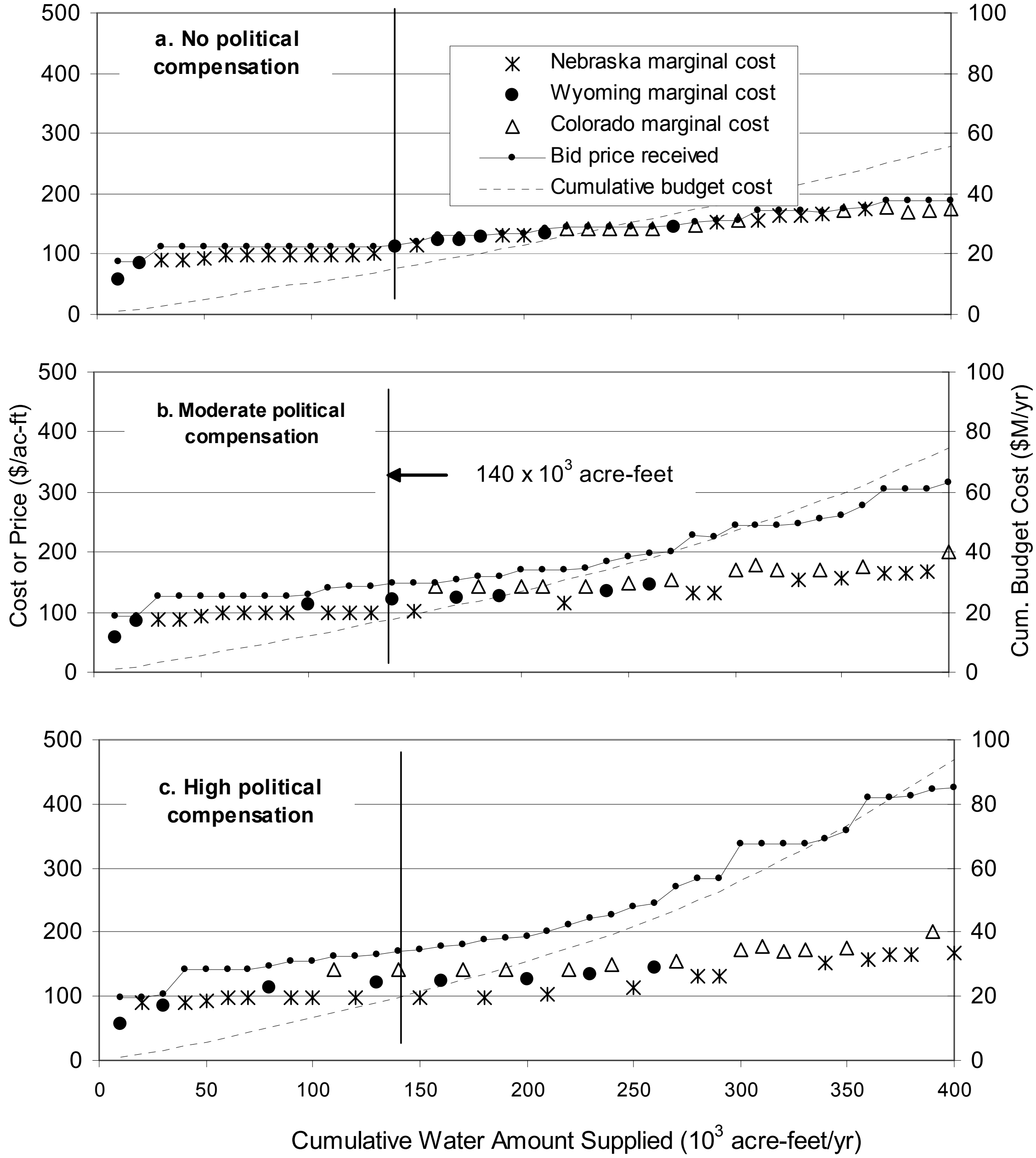

FIGURE 6-2

Price of 10,000-acre-foot increments of environmental water, and cumulative cost, assuming different levels of political compensation

TABLE 6-6

Welfare effects from supplying 140,000 acre-feet of environmental water.

Level of Political Compensation	Welfare Costs[a]				
	Colorado	Nebraska	Wyoming	Federal	Total
None					
Water Supplied (AF/yr)	0	110,000	30,000	0	140,000
Budget Cost ($/yr)	-3,057,823	-3,057,803	-1,528,912	-7,644,560	-15,289,120
Second Price Gain	0	+1,772,443	+360,652	0	+2,133,095
Political Compensation	0	0	0	0	0
Net Welfare[a]	-3,057,823	-1,285,360	-1,168,260	-7,644,560	-13,156,003
Moderate					
Water Supplied (AF/yr)	0	100,000	40,000	0	140,000
Budget Cost ($/yr)	-3,552,900	-3,552,900	-1,776,450	-8,882,250	-17,764,500
Second Price Gain	0	+1,362,207	+404,612	0	+1,776,819
Political Compensation	0	+2,201,000	+442,700	0	+2,643,700
Net Welfare	-3,552,900	+10,307	-929,138	-8,882,250	-13,353,981
High					
Water Supplied (AF/yr)	20,000	80,000	40,000	0	140,000
Budget Cost ($/yr)	-3,960,740	-3,960,740	-1,980,370	-9,901,850	-19,803,700
Second Price Gain	+111,191	+867,424	+469,561	0	+1,448,176
Political Compensation	+372,900	+2,881,100	+889,500	0	+4,143,500
Net Welfare	-3,476,649	-212,216	-621,309	-9,901,850	-14,212,024

[a]Welfare costs represent the real cost of the water to all parties combined. Net welfare is equal to the budget cost less that part of the budget cost which represents transfer payments. Both second-price gain and political compensation payments affect the distribution of welfare among the parties but not total welfare, because the loss to the paying party equals the gain to the receiving party.

compared to a least-cost scenario). The auction approach would resolve principal-agent problems (see Section 2.5.5) by creating incentives for each state to incrementally reveal its true political compensation costs. The resulting agreement is likely to benefit all the parties because each can choose between supplying the water at an acceptable minimum price or paying someone else to supply it.

6.3.2 Model II: Determining how much water to allocate to environmental use

Whereas Model I examined only the provision of water and constrained the problem to a negotiation among three States, Model II casts the negotiation problem more broadly. Questions (policy attributes) examined in this model were the following:

1. What method or approach should be used for meeting endangered species' needs in the central Platte River floodplain (*Method* attribute)?
2. What is the appropriate level of investment in meeting species' needs (*Cost* attribute)?
3. Who should make that investment (*Who pays* attribute)?

The players included the federal government and environmental and agricultural interest groups, as well as the states. Because all parties stand to gain if agreement is reached, the decision process was modeled as a cooperative multilateral bargaining game. Policy options were defined as a combination of the three policy attributes. For each attribute there were five choices or levels , i.e., five methods, five cost alternatives and five payment policies, which produced a potential for 125 different policies ($5^3 = 125$). Policy evaluation criteria were based on the utility of (i.e., relative preference for) each policy on the part of each of the game participants. Utility was also expected to vary not only by group but also according to the level of knowledge about ecological risks and the likely regional impacts of environmental policies.

To develop this game, it was necessary to conduct a survey of preferences in Nebraska, Colorado and Wyoming. The following subsections will discuss, respectively, the survey approach, the mathematical definition of Model II, and the results of Model II simulations.

6.3.2.1 Household survey of environmental preferences

In November 2000, a total of 4,150 households in Colorado, Nebraska and Wyoming were randomly selected from lists compiled by Experian (Costa Mesa, CA), a private company specializing in the compilation of mailing lists. Survey procedures consisted of a first mailing, followed by a reminder postcard about 10 days later; then those who had not responded within 10 days following the postcard were sent a second copy of the survey.

The survey consisted of four parts. In Parts 1-3, respondents were posed a series of statements and asked to indicate whether they agreed or disagreed (Parts 1 and 2) or opposed or supported (Part 3) each statement, on a five-point scale.[61] Part 1 assessed general attitudes regarding water and threatened and endangered species policy in the three Platte River states, but because these responses did not figure directly in model construction they are not discussed in detail here. Part 2 examined technical beliefs and the responses were used to assess the effect of respondent level of knowledge on policy preferences. Part 3 examined policy attributes and options, and these responses were used to compute respondent and interest group preferences for various policy attributes. Part 4 asked questions about demographics, and this information was used to identify respondents with particular bargaining groups (state of residence, and agricultural or environmental interest group) to be represented in the model.

6.3.2.1.1 Level of knowledge

The 10 statements posed to respondents in Part 2 (Table 6-7) were similar in form to risk hypotheses, which postulate a causal relationship between a source or a stressor and an endpoint.

TABLE 6-7

Statements used in the household preferences survey to assess respondent level of knowledge; answers regarded by researchers as correct; and basis. Respondents were asked to rate agreement/disagreement on a five point scale.

Technical statement appearing in Part 2 of household preference survey	Correct answer	Basis for statement/answer (and relationship to risk hypotheses as numbered in Table 6.1)
a. Maintaining a wider Platte River channel is not necessary for sustaining a large and healthy Sandhill Crane population.	False	Cadmus Group;[52] Currier & Ziewitz[29] (Risk Hypotheses 1 & 7)
b. Increased stream flow will help maintain a wide Platte River channel for use by cranes and other wildlife.	True	Sidle et al..;[15] McDonald & Sidle, [62] (Risk Hypotheses 1 & 7)
c. Increased wet meadow acreage is needed to meet the food needs of cranes and other wildlife in the Central Platte Valley.	True	Cadmus Group;[52] Faanes & LeValley;[24] Currier & Ziewitz[29] (Risk Hypotheses 9 & 11)
d. Increased instream flows would significantly increase the quantity and quality of wet meadows.	True	Hurr;[63] *The Groundwater Atlas of Nebraska*[64] (Risk Hypotheses 5 & 11)
e. The changes in regional income and employment that result from reallocating up to 420,000 acre-feet of water from agriculture to endangered species are likely to be so small that they will go unnoticed by most of the people living in the Platte Valley region.	unknown	
f. Policies to maintain or increase the current flows in the Platte River will lead to increased water costs for people living in communities located near the river.	unknown	
g. Ground water irrigation has lowered the water table in some parts of the Central Platte Valley.	True	*The Groundwater Atlas of Nebraska*[64]
h. Ground water irrigation has adversely affected wet meadows in some parts of the Central Platte Valley.	True	*The Groundwater Atlas of Nebraska*[64]
i. Improved habitat will result in an increased number of Sandhill Cranes using the Platte River.	unknown	
j. An increased number of Sandhill Cranes will result in increased tourism in the Central Platte region.	unknown	

Whereas risk hypotheses generally refer to existing relationships, however, these statements tended to be in the form of inferences about the future, to emphasize their relevance to policy. Six of the 10 statements are regarded to have correct answers; the other four were of interest because they are often claimed, but their veracity is uncertain. Seven of the 10 statements pertained to ecological endpoints, including the shallow water table, wet meadows, cranes and other wildlife. These statements roughly corresponded to several of the risk hypotheses (Tables 6-5 and 6-7).[a] Three of the seven ecological statements dealing with the habitat needs of cranes were based on the expert opinion of the researchers. A simple sum of responses to the six verifiable statements constituted the knowledge index, KL, in Model II, after appropriate transformations so that a higher value meant more knowledge in all cases.

6.3.2.1.2 Utility of policy attributes

In Part 3, each of the three policy attributes, *Method*, *Cost* and *Who pays*, was described, and five different levels were defined for each (Table 6-8). Respondents were asked to rate their support of each of these 15 attribute levels individually. Next, seven policy options (each consisting of one *Method* level, one *Cost* level and one *Who pays* level) were selected out of the total of 125 possible combinations that would capture the range of potential responses over each attribute. Utility ratings for these options were used to derive attribute weights in Model II as further described below. These attribute weights were multiplied times utility scores for each attribute and summed across the three attributes that define a policy to determine the utility scores for all 125 policy options.

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[a] The fact that many of those hypotheses were not evaluated in the W-ERA does not mean that these statements are not scientifically supported; in many cases the hypothesis is regarded as supported but the underlying relationship needs to be better quantified.

TABLE 6-8

Descriptions of the three policy attributes and their respective levels, a-e, that were evaluated in part 3 of the household preferences survey

Method:	Five different methods for meeting threatened and endangered species needs on the central Platte are described below. a. Meet all endangered species needs using least cost methods of water conservation, water reallocation and riparian land management, even if this means purchasing or leasing substantial quantities of water from agriculture. b. Meet all endangered species needs using a combination of water conservation, water reallocation and riparian land management programs, but minimize the purchase or leasing of water from agriculture, even if this increases the cost of meeting these needs. c. Meet as many endangered species needs as possible using riparian land management and water conservation programs to provide for endangered species, but do not purchase or lease any additional water from agriculture, even if this means that the continued existence of the species involved may be at risk. d. Use a combination of water conservation, water reallocation and riparian land management implemented on a trial basis over several years to make certain that the program is necessary and effective before making large public investments, even if this means there is a potential for continued risk to threatened and endangered species. e. Invest in all endangered species protection methods as long as the economic benefits from such investments are greater than the costs, even if this means continued risk to threatened or endangered species.
Cost:	To provide for threatened and endangered species on the Platte River, the cost to federal taxpayers throughout the U.S. and state taxpayers in Colorado, Nebraska and Wyoming could range from zero to $40,000,000 per year. The amount will depend on what priority we choose to attach to species protection; on the level of risk to species extinction that we choose to accept; and on the species protection methods that we choose to use. Five different investment policies for meeting threatened and endangered species needs on the central Platte are described below. a. Invest nothing to protect Whooping Cranes, Least Terns and Piping Plovers. b. Invest whatever the U.S. Fish and Wildlife Service (USFWS) says is needed for the species to return to non-threatened status (currently estimated to cost as much as $40,000,000 per year). c. Invest about 25 percent of what the USFWS says is needed, or $10,000,000 per year. d. Invest about 50 percent of what the USFWS says is needed, or $20,000,000 per year. e. Invest about 75 percent of what the USFWS says is needed, or $30,000,000 per year.
Who pays:	Another important policy dimension concerns the question of who should pay for species protection. Should it be the federal government, the states involved in using the resources, private environmental interests, or some combination? The following five potential policies reflect these choices. a. All costs paid by the federal government. b. Federal government pays 50 percent and private environmental interests pay the remaining 50 percent. c. Federal government pays 50 percent and the states of Colorado, Nebraska and Wyoming pay equal shares of the remaining 50 percent. d. Federal government pays 50 percent and the states of Colorado, Nebraska and Wyoming pay the remaining 50 percent in proportion to the amount of Platte River water consumed in each state (Colorado 20%, Nebraska 20% and Wyoming 10%). e. Federal government pays one-third, private environmental interests pay one-third and the states of Colorado, Nebraska and Wyoming split the remaining one-third in proportion to the amount of Platte River water consumed in each state (Colorado 13%, Nebraska 13% and Wyoming 7%).

The utility of a given environmental policy, within a particular interest group, was defined as the adjusted sum of preference scores for the attributes of that policy, as follows:

$$U_{ij} = W_{i1} M_{ij} + W_{i2} C_{ij} + W_{i3} P_{ij} + KAF_{ij} \qquad 6\text{-}3$$

where:

U_{ij}	=	utility or preference score for interest group *i*, policy option *j*;
M_{ij}	=	attribute score by interest group *i* for *Method*, policy *j*, on a 1 to 5 scale;
C_{ij}	=	attribute score by interest group *i* for *Cost*, policy *j*, on a 1 to 5 scale;
P_{ij}	=	attribute score by interest group *i* for *Who pays*, policy *j*, on a 1 to 5 scale;
KAF_{ij}	=	knowledge adjustment factor for interest group *i*, policy *j*, as described below;

and W_{i1}, W_{i2}, and W_{i3} are attribute weights. The knowledge adjustment factor (KAF) was defined as the difference between the mean U_{ij} for those in interest group *i* whose knowledge level KL_i, as defined in Section 6.3.2.1.1, was one standard deviation or more above the mean and the mean U_{ij} for the entire interest group *i*. However, KAF was set to zero unless the participants in the game chose to invest in education as one method of reaching agreement, or chose to ignore the preferences of those in each interest group who were not technically knowledgeable.

The attribute weights W_{i1}, W_{i2} and W_{i3} would be unnecessary if *Method*, *Cost* and *Who pays* were of equal importance to respondents within a given interest group. If this were the case, then the overall utility U_{ij} of a policy option (after adjusting to equivalent scales) would be similar whether it was derived by summing a group's mean utility scores for the individual attribute levels that composed the policy or using that group's utility scores for the policy evaluated as a whole. Because this was not the case, raw attribute weights B_1, B_2 and B_3 were determined for each interest group *i* by regressing raw utilities RU_i for the seven whole policies over the scores of the three individual attributes to obtain the following equation for each group:

$$RU_i = B_{io} + B_{i1}M_i + B_{i2}C_i + B_{i3}P_i + \varepsilon_i \qquad 6\text{-}4$$

where M, C and P are the 1 to 5 scores for the three policy attributes and ε is an error term. The regression coefficients were then normalized across attributes to get a total value of 1.0 by dividing each non-normalized "B_i" value by the quantity ($B_{i1} + B_{i2} + B_{i3}$), such that for each group the normalized weights become:

$$W_{i1} + W_{i2} + W_{i3} = 1.0 \qquad 6\text{-}5$$

These normalized weights were then used to adjust the individual attribute scores for all 125 policy alternatives as shown in Equation 6-3.[a]

6.3.2.2 Bargaining theory and model solutions

The previous subsection defined utility for each policy by bargaining group. Here the problem of combining those utilities to identify the most globally preferred policies is addressed. The primary objective of the bargaining process is to find the policy option, defined as a combination of policy attributes, that maximizes total utility and is acceptable to all groups. In the bargaining literature and the broader literature of social and public choice, certain solution concepts seem to prevail. This section will introduce three of the most commonly used solution concepts for the bargaining model at hand: the utilitarian, Nash and egalitarian solutions. Each of these solutions will later be applied to the data obtained from the survey to determine if there are policy options that emerge repeatedly. An option chosen by different bargaining processes,

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[a] The concept of utility as used here is simply a preference rating. It depends on how important the consequences of a policy choice are to the respondent and also on what he or she believes the consequences will be. Knowledge can influence utility by changing the respondents' beliefs regarding consequences.

which represent different social judgments, is most likely to be the policy option that would emerge from a real bargaining game. If, on the other hand, the policy options chosen by different bargaining solutions are very different, then one has to investigate the conditions of the bargaining process and the background for the social judgment much more carefully. If there is no attribute-level combination that is minimally acceptable to all groups, the players have four options: 1) negotiate a lower level of minimally acceptable utility; 2) change the water supply costs by negotiating a reduction in the political compensation factor in Model I; 3) change the preference functions of participants by providing improved biological and/or economic information; or 4) declare an infeasible solution.

Let X denote the set of available alternatives. In our case, X equals the set of 125 policy options that could be chosen. Let N denote the set of agents. Later on, three different sets of agents will be considered:

N={C,N,W},

N={Agricultural Interest, Environmental Interest}={Ag,En}, and

N={AgColorado, AgNebraska, AgWyoming, EnColorado, EnNebraska, EnWyoming}

={AgC, AgN, AgW, EnC, EnN, EnW}.

To model the theory applied to these agents, a generic set of agents N={1,…,n} and a generic agent *i* are denoted. Similarly, there are a generic set of alternatives X and generic alternatives x and y. Next, it is assumed that each agent associates a cardinal utility u_i (x) with each policy option x, estimated as u_{ij} in Equation 6-3. (Alternatively, the ordinal ranks of alternatives are taken as utility information, ignoring intensities of utility across alternatives and across agents.) Since now each policy option x induces a vector (u_1 (x),…, u_n (x)), the decision of choosing a policy option boils down to deciding which vector of utilities is acceptable for all agents. In order to preserve efficiency of bargaining outcomes, only bargaining solutions that are Pareto efficient are considered; that is, for any policy option chosen by the bargaining solution, there does not exist another policy option such that all agents are weakly better off and at least one agent is strictly better off (see Section 2.2.1).

6.3.2.2.1 Utilitarian solution

The utilitarian solution is the policy option which maximizes the sum of all agents' utilities and can be depicted as

$$\max_{x \in X} \sum_{i=1}^{n} u_i(x) \qquad \text{6-6}$$

where: u_i is the cardinal or ordinal utility for agent *i*, for some vector of policy options x.

6.3.2.2.2 Nash solution

The Nash solution is the policy option which maximizes the product of all agents' utilities and can be depicted as

$$\max_{x \in X} \prod_{i=1}^{n} u_i(x) \qquad \text{6-7}$$

with all terms as defined previously.

6.3.2.2.3 Egalitarian solution

The egalitarian solution is the Pareto efficient policy option which minimizes the sum of the differences between all agents' utilities. Mathematically this solution can be defined as

$$\min_{x \in X} \sqrt{\sum_{i=1}^{n} [\frac{\sum_{j=1}^{n} u_j(x)}{n} - u_i(x)]^2} \qquad \text{6-8}$$

where: u_i is the cardinal or ordinal utility for agent *i*, and u_j is that for all other agents, for some vector of policy attributes x.

In terms of social policy, the utilitarian solution represents that set of decision rules where there is no concern for the relative utility of agents. Any gain in total utility is considered an improvement irrespective of how the total is distributed across agents. The Nash solution essentially incorporates the concept of diminishing marginal utility, while the egalitarian solution takes the potential concern for equity or fairness one step further. Let us demonstrate with a simple example. Suppose there are two agents and three policy options. Option A produces 1 unit of utility for agent 1 and 10 units for agent 2; option B produces 4 units for each agent; and option C produces 6 units for agent 1 and 3 units for agent 2. In this case, the utilitarian solution would favor option A ($1+10 > 6+3 > 4+4$), whereas the Nash solution would favor option C ($6 \times 3 > 4 \times 4 > 1 \times 10$), and the egalitarian solution would favor option B ($4-4 < 6-3 < 10-1$). The respective solutions can also be referred to as the sum, product and equity solutions.

6.3.2.3 Survey results

This section summarizes the survey findings with an emphasis on their application to model calculations; tabularized responses to survey questions are presented in Supalla et al.[65] A total of 1,187 useable surveys were returned, for an overall response rate of 26 percent. The response rate for Nebraska residents was highest at 32 percent, followed by Wyoming at 24 percent and Colorado at 22 percent. These relatively low response rates suggest a likelihood of response bias, although there were no particular indications of response biases within or between interest groups. One would generally expect, however, that those who were better educated and most interested in the problem would be the most likely to respond.

6.3.2.3.1 Demographics

Demographic responses showed that respondents were in fact somewhat older and better educated than the general population. The average age of respondents was 53, over 38 percent had a Bachelor's degree or better education, and less than 5 percent had not graduated from high school. The age distribution was essentially the same for each state, but the Colorado respondents were significantly better educated than those from Nebraska or Wyoming. Approximately 14 percent of respondents were farmers or ranchers, over 18 percent were self employed in other ways, about 13 percent worked for state or local government and the remainder were either employed by other types of organizations or retired. The employment distribution was very similar for each state, except for agriculture. Very few of the Colorado respondents were farmers or ranchers (7%), compared to 12 percent for Wyoming and 19 percent for Nebraska. Differences in the proportion of state respondents who were farmers or ranchers reflect in part, actual differences in the proportion of each state's population that is engaged in agriculture, but these differences may also reflect a self-selection bias. Farmers in Nebraska, especially central Platte irrigators, are more likely to be directly impacted by central Platte programs and, thus, more likely to take the time to respond to the survey.

A relatively large number of respondents were affiliated with agricultural or environmental interest groups. In total, about 17 percent of respondents were affiliated with agricultural groups and 31 percent with environmental groups. The three states were quite similar, except that only 8 percent of Colorado respondents were affiliated with agriculture, and only 19 percent of Nebraska respondents were affiliated with environmental groups compared to 48 percent in Colorado. This suggests that interest groups may be a major source of information on central Platte issues for that part of the population that was interested enough in the issues to respond to the survey.

6.3.2.3.2 Attitudes regarding environmental policy

About one-third of respondents agreed that society should ensure species protection regardless of cost. There was very strong support for having the federal government and private environmental organizations pay for species protection rather than the states. Two-thirds of respondents agreed that the federal government rather than the states should pay for most of the cost and 80 percent agreed that private environmental organizations should also contribute. There was also strong support for the idea that the economic base provided by irrigated agriculture should be protected. Over 70 percent would be willing to pay more for species protection to protect the economic base, and over 50 percent were willing to protect the economic base even if it meant increased risk to endangered species. Surprisingly, 55 percent would support paying twice as much for environmental water as an alternative to reducing irrigation.

There were few significant attitudinal differences between the states. Colorado residents were much more likely than Wyoming or Nebraska residents to agree that society should ensure environmental integrity regardless of the cost. Wyoming respondents were not supportive of each state supplying one-third of the environmental water, while Nebraska respondents supported this alternative. This probably reflects a concern among Nebraska residents that the state may be asked to provide more than a one-third share and a belief by Wyoming residents that their equitable share is less than one-third.

6.3.2.3.3 Technical beliefs regarding central Platte River environmental problems

There was considerable disagreement and/or lack of knowledge concerning physical environmental attributes. Only 24 percent of Colorado residents, 29 percent of Wyoming residents and 41 percent of Nebraska residents were aware, defined as agreed or strongly agreed, that a wide river channel is important to cranes. Less than 50 percent of the respondents in all states recognized that increased stream flow would help maintain a wide river channel. There was greater recognition of the environmental importance of wet meadows and of the link between groundwater irrigation and wet meadow production, but the number of correct responses was still below 50 percent in nearly all cases. Respondents in all states also expressed considerable uncertainty with respect to the economic effects from management alternatives. Nearly an equal number of people agreed as disagreed with statements concerning the effects of changes in the amount of irrigation or tourism on the regional economy.

Differences between the states may suggest some reasons for the technical beliefs that are held. Over 21 percent of Nebraska respondents disagreed with the statement that groundwater irrigation adversely affects wet meadows, compared to 11 percent for Colorado and 12 percent for Wyoming. Similarly, 22 percent of Nebraska respondents disagree with the contention that improved habitat will increase the number of cranes, compared to 10 and 13 percent for Colorado and Wyoming, respectively. These differences suggest that there may be an inclination on the part of some respondents to deny recognition of technical relationships that do not support their policy position and/or that imply some responsibility for an adverse impact. The Nebraska sample contains a relatively large proportion of irrigators, many of whom may be reluctant to accept scientific claims about how their activities may affect the Middle Platte ecosystem.

6.3.2.3.4 Level of support for policy attributes

Data on the level of public support for each of five different levels of each of three policy attributes were used in game models to find bargained policy solutions. Preferences were analyzed by state and for each of two interest-related bargaining groups, agricultural and environmental (Table 6-9). Respondents were classified as agricultural if they indicated that they were self-employed as a farmer or rancher, employed by an agricultural interest group or affiliated with the Farm Bureau, the Farmers Union or an irrigation district. Respondents were classified as environmental if they indicated that they were employed by an environmental interest group; affiliated with the Sierra Club, The Nature Conservancy or the Audubon Society; or agreed or strongly agreed with the statement that "Society should ensure that the needs of threatened and endangered species are met regardless of economic cost." Respondents who qualified as agricultural based on employment or interest group affiliation, but who also agreed that society should meet the needs of endangered species irrespective of economic cost, were considered as both agricultural and environmental. Those respondents who either could not be classified as exclusively agricultural nor exclusively environmental were classed as "other" and were included in state totals but were not analyzed as a separate bargaining group.

For the *Method* attribute, the level receiving the strongest support from all states as measured by the average score for all residents was adaptive management (Appendix 6-A). Colorado's second best choice was to meet all needs while minimizing water, but the second best option preferred by Nebraska and Wyoming respondents was to do the best possible job of meeting endangered species needs with no reallocation of irrigation water. Agricultural interests in all states strongly preferred either an adaptive management approach or a program that produced as much endangered species protection as possible without reallocating any water from

TABLE 6-9

Respondent classification into bargaining groups, by state. Based on type of employment, interest-group affiliation, and attitude regarding endangered species, a respondent could be classified as either agriculture, environmental, both, or neither.

Bargaining Group	Colorado	Nebraska	Wyoming	All States
	Numbers of Respondents			
Agriculture	24	105	55	184
Environmental	143	86	110	339
Other	132	257	166	555
Total	299	448	331	1,078

agriculture (Appendix 6-A). They were most strongly opposed to the idea of meeting all needs irrespective of the costs. Environmental interests preferred to meet all needs, although they also expressed considerable support for an adaptive management approach.

Expressed support for different levels of investment (*Cost* attribute) was somewhat mixed, but the strongest support in all states was for a $10M annual investment, which is about 25 percent of what many observers believe it would take to fully implement USFWS recommendations. However, 32 percent of all Colorado respondents expressed strong support for investing whatever it took to meet USFWS recommendations. Agricultural interests preferred to invest nothing, or perhaps $10M per year, but there was very little support among agriculturalists in all states for spending more than $10M per year.

The payment policy results (*Who pays* attribute) were especially interesting. All states preferred that private environmental groups pay a significant part of the cost, which is contrary to current proposals to address the problem. The reasons for preferring private contributions are unknown, but the leading hypothesis is that respondents believe those who get the most utility from environmental improvements should also pay the most. The first choice of all states was a payment policy consisting of one-third federal, one-third private and one-third state, with the state one-third being distributed between the three states in proportion to current water use. Wyoming respondents objected strongly to each state paying a equal share of the aggregate state share, but there were no other significant differences between the states. The strongest support for some private contribution to the cost of meeting endangered species needs came from agricultural interests, but surprisingly there was also substantial support from environmental interests for requiring some private cost sharing. This may reflect a belief that the benefits from endangered species protection accrue disproportionately to environmental interests and, thus, the entire burden should not fall to general taxpayers.

6.3.2.4 Model II results

6.3.2.4.1 Weights for policy attributes

Responses to a sampling of 7 of the 125 policies were used as described in Equations 6-4 and 6-5 to derive attribute weights for each of the three states and for agricultural and environmental bargaining groups within each state (Appendix 6-A). Except for the environmental interest group in Wyoming, the most heavily weighted policy attribute was payment policy and the least important was the method of meeting endangered species needs. Environmental interests generally placed more weight on method and less on payment policy, compared to agricultural interest groups.

6.3.2.4.2 Policy preferences

Weighted utility scores were computed for all 125 policy options for each bargaining group using Equation 6-3; these are cardinal utilities (not presented). To facilitate comparisons, utility scores for each group were ranked from 1 to 125, where the best option is ranked 125 and the poorest has a ranked score of one; these are ordinal utilities. The full array of 125 policy options was then reduced to 17 by eliminating those which were not Pareto efficient (Tables 6-10, 6-11 and 6-12). An option was considered Pareto inefficient if it was possible to improve the level of total utility across groups without making one or more groups worse off. The level of support for the more efficient options was considered in more detail.

Surprisingly, the highest ranked option in each state was the same, option N, which consists of an adaptive management program using both riparian land management and improved stream flow to protect endangered species, at an investment level of $10M per year, with the

TABLE 6-10

Definition of Pareto efficient policy options: attribute levels corresponding to each policy.

Policy Option	Attribute Level[a]		
	Method	*Cost*	*Who pays*
A	d. Adaptive Management	c. Invest $10M, 25% of Need	a. All Costs Paid by Feds
B	d. Adaptive Management	a. Invest Nothing	b. Feds 50%, Private 50%
C	d. Adaptive Management	b. Invest $40M, per USFWS	b. Feds 50%, Private 50%
D	a. All Needs, Least Cost	c. Invest $10M, 25% of Need	b. Feds 50%, Private 50%
E	d. Adaptive Management	c. Invest $10M, 25% of Need	b. Feds 50%, Private 50%
F	a. All Needs, Least Cost	d. Invest $20M, 50% of Need	b. Feds 50%, Private 50%
G	e. Benefit-Cost Approach	c. Invest $10M, 25% of Need	d. Feds 50%, States 50% Proportional to Use
H	d. Adaptive Management	a. Invest Nothing	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
I	a. All Needs, Least Cost	b. Invest $40M, per USFWS	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
J	b. All Needs, Minimum Water	b. Invest $40M, per USFWS	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
K	d. Adaptive Management	b. Invest $40M, per USFWS	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
L	a. All Needs, Least Cost	c. Invest $10M, 25% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
M	b. All Needs, Minimum Water	c. Invest $10M, 25% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
N	d. Adaptive Management	c. Invest $10M, 25% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
P	a. All Needs, Least Cost	d. Invest $20M, 50% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
Q	b. All Needs, Minimum Water	d. Invest $20M, 50% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use
R	b. All Needs, Minimum Water	e. Invest $30M, 75% of Need	e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use

[a] A full description of each policy attribute and level is found in Table 6-8.

TABLE 6-11

Pareto efficient policy preferences,[a] by state

Policy Option[b]	Ranked Utility Scores		
	Colorado	Nebraska	Wyoming
A	78	104	89
B	80	89	110
C	93	84	103
D	70	94	91
E	98	117	119
F	65	69	77
G	69	82	60
H	117	119	123
I	113	92	92
J	119	99	104
K	123	114	121
L	116	120	114
M	121	122	120
N	125	125	125
P	115	105	96
Q	120	113	113
R	118	102	101

[a] Policy options are ranked from 1 to 125 with 125 being the highest or best option.
[b] See Table 6-10 for a description of each policy option.

TABLE 6-12

Pareto efficient policy preferences,[a] by bargaining group and state

Policy Option[b]	Colorado		Nebraska		Wyoming		All Ag Utility Rank	All Envl. Utility Rank
	Ag Utility Rank	Envl. Utility Rank	Ag Utility Rank	Envl. Utility Rank	Ag Utility Rank	Envl. Utility Rank		
A	115	42	94	76	111	46	105	42
B	116	8	125	28	125	27	125	9
C	53	82	106	55	106	73	98	71
D	113	46	114	39	113	68	114	47
E	124	43	122	59	123	55	123	45
F	63	62	103	30	93	69	92	53
G	90	58	47	83	46	85	50	79
H	117	54	112	113	107	94	115	91
I	33	123	51	115	42	125	45	124
J	44	125	60	120	47	124	52	125
K	52	122	72	122	65	123	72	123
L	114	109	84	117	73	121	84	115
M	120	117	93	124	82	119	94	120
N	125	105	108	125	97	115	110	114
P	64	115	71	114	51	122	65	117
Q	75	121	77	118	60	120	74	121
R	58	119	61	121	49	117	56	122

[a] Policy options are ranked from 1 to 125 with 125 being the highest or best option.
[b] See Table 6-10 for a description of each policy option.

federal government paying one-third, the states one-third and private environmental groups one-third (Table 6-11). Under this option the states' share is split proportionally between the states according to historical water use. The lowest ranked options in all states were generally those which called for investing nothing.

Policy preferences of interest groups within a state were much more varied (Table 6-12). The first choice of agricultural interests in both Nebraska and Wyoming was option B, which consists of adaptive management at a very low level of investment, with all costs paid by the federal government and private environmental interests. Agricultural interests in Colorado preferred option N, which is surprisingly consistent with the preferences of all citizens in each of the three states. Environmental interests in Colorado and Wyoming preferred meeting all endangered species needs, while reallocating as little water as possible, with expenditures up to $40M per year, with costs shared equally by the federal government, the states and private interests.

6.3.2.4.3 Bargaining solutions

The bargaining challenge, therefore, lies in finding a solution to differences of opinion within, rather than between, states. The magnitude of this challenge can be seen by analyzing how acceptable a given group's preferred option is to competing bargaining groups (Table 6-13). For example, examining the seventh row of Table 6-13, all agriculture prefers an adaptive management plan with minimal water reallocation and minimal investment, with 50 percent of the costs paid by private environmental groups and 50 percent by the federal government (option B). Moving to the end of the seventh row, environmental interests aggregated across states rank option B as their ninth poorest option, which places it in the bottom 10 percent of the 125 choices being considered. Environmental interests (last row) prefer option J, which would meet

TABLE 6-13

Comparison of preferred policy options between competing interest groups

Group	Preferred Option	CO	NE	WY	CO Ag	NE Ag	WY Ag	All Ag	CO Envl	NE Envl	WY Envl	All Envl
		Rank of Preferred Option[a]										
CO	N	**125**	125	125	125	108	97	110	105	125	115	114
NE	N	125	**125**	125	125	108	97	110	105	125	115	114
WY	N	125	125	**125**	125	108	97	110	105	125	115	114
CO Ag	N	125	125	125	**125**	108	97	110	105	125	115	114
NE Ag	B	80	89	110	116	**125**	125	125	8	28	27	9
WY Ag	B	80	89	110	116	125	**125**	125	8	28	27	9
All Ag	B	80	89	110	116	125	125	**125**	8	28	27	9
CO Envl	J	119	99	104	44	60	47	52	**125**	120	124	125
NE Envl	N	125	125	125	125	108	97	110	105	**125**	115	114
WY Envl	I	113	92	92	33	51	42	45	123	115	**125**	124
All Envl	J	123	40	108	12	49	104	3	125	125	92	**125**

[a] Policy options are ranked from 1 to 125 with 125 being the highest or best option

all endangered species needs at a cost of up to $40M per year, with costs shared equally between the federal government, the states and private environmental interests. Agricultural interests rank

option J as their third poorest option. These comparisons suggest that a bargaining process is needed to find an acceptable middle ground that lies somewhere between, at one extreme, a program that meets all endangered species needs (as determined by USFWS), involves a major reallocation of water from agriculture, and costs up to $40M per year; and at the other extreme, a program that reduces the reallocation of water to an absolute minimum, costs much less, but exposes endangered species to significant risk.

Three solutions to a multilateral bargaining game were computed in a search for the policy options most likely to be acceptable to all of the principal interest groups (Table 6-14). Policy N is both the utilitarian and Nash solution (Equations 6-6 and 6-7), whether using cardinal or ordinal utility. However, the egalitarian solution (Equation 6-8) is policy option D when using cardinal utility and option A when using ordinal utility.

These results suggest that if the bargaining agents were not concerned about equity between groups they would adopt policy N, which is an adaptive management approach meeting only some of the endangered species needs, spending $10M per year, with the costs split evenly between the federal government, the states and private environmental groups. However, if equity was more of a concern, the solution would involve a similar approach with about the same level of investment, but with no state contribution to program costs.

If policy option N is selected, environmental groups are likely to be reasonably satisfied, because a reasonable amount of endangered species protection will be provided and the costs will be widely shared. However, at least part of the agricultural community is likely to be

TABLE 6-14 Results of bargaining models, all bargaining groups						
Pareto Efficient Options	Cardinal Utility			Ordinal Utility		
	Utilitarian	Nash	Egalitarian	Utilitarian	Nash	Egalitarian
	Rank of Policy Option [a]					
A	101	104	123	98	95	121
B	102	103	87	85	56	106
C	87	87	73	92	98	105
D	105	109	125	102	96	120
E	116	117	118	110	105	94
F	81	82	90	82	80	40
G	72	69	63	76	81	88
H	122	122	95	121	119	25
I	103	91	8	99	90	2
J	111	105	9	107	103	98
K	115	112	20	115	114	11
L	123	123	49	123	123	44
M	124	124	32	124	124	1
N	125	125	70	125	125	91
P	110	108	24	112	111	4
Q	117	115	29	117	117	78
R	109	106	19	109	109	101

[a] Options are marked from 1 to 125, with 125 being most preferred.

uncomfortable with a program that reallocates water away from agriculture in ways they believe may not be justified on a cost-benefit basis, especially when the states are paying a significant share of the cost.

6.3.2.4.4 Potential impact of education on policy preferences

An important policy issue concerns the extent to which education might reduce the level of disagreement between bargaining groups. Two questions would need to be answered. First, does the tendency for groups to disagree appear to be related to the level of technical knowledge within the groups? If the answer is yes, then would education improve the level of technical knowledge and the level of agreement? While the second question was beyond the scope of the current project, the first question was analyzed by comparing the policy preferences of more and less knowledgeable survey respondents.[a]

Knowledgeable respondents were defined as those whose knowledge index score, as defined in Section 6.3.2.1.1, was at least one standard deviation above the mean in each state. Average utility scores for the knowledgeable and non-knowledgeable classes were computed and compared for the 17 Pareto efficient policy options. An aggressive education program was arbitrarily assumed to be able to change the level of support for the Pareto efficient policies by non-knowledgeable citizens by an amount equal to one-half the average difference between the knowledgeable and non-knowledgeable classes. Hence, the appropriate adjustments were made to the non-knowledgeable scores and a new interest group average calculated for each Pareto efficient policy option. Rank orderings of the 17 options with and without the assumed education

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[a] The effect of knowledge on policy preferences was also addressed with a logit model which analyzed the effect of knowledge on the probability that an individual would support environmentally intense policies. This analysis found a strong statistical relationship between knowledge and level of policy support.

effect were then compared to determine if there was any appreciable effect on what option was most preferred by each interest group and, most importantly, to determine if the knowledge effect brought the interest groups closer to an agreement on the best policy option.[65]

In all states the effect of improved knowledge was to bring the agricultural and environmental interest groups closer to agreement. In Nebraska, the effect was primarily on the agricultural interest group. Nebraska agriculture's first choice went from option B, which calls for investing nothing in endangered species protection, to option N, which was the first choice of Nebraska environmental interests before the effect of improved knowledge. With improved knowledge the first choice of Nebraska environmental interests became option J, which is similar to option N, but calls for a higher level of investment. For Wyoming, the effect of improved knowledge was also to make environmentally strong options more acceptable to agricultural interests. Both Wyoming agricultural and Wyoming environmental interests preferred option I after the knowledge effect was imposed, whereas previously, Wyoming agricultural interests preferred a much lower level of investment in endangered species protection. For Colorado, there was no significant knowledge effect on environmental interests, but agricultural preferences changed from preferring adaptive management option N to preferring to meet all needs, option L.

6.3.2.4.5 Policy implications of Model II

The results from Model II suggest that the most important differences of opinion regarding central Platte management policies exist between agricultural and environmental interest groups within each state, rather than between states. At the aggregate level, all three states preferred a policy which called for an adaptive management approach that minimized the reallocation of water from agriculture and involved a modest level of investment, with the costs shared equally between the federal government, the states and private environmental interests. Within Nebraska and Colorado, however, agricultural interests preferred to invest nothing, with everything paid for by the states and private environmental interests, while environmental interests preferred a much more aggressive program to ensure endangered species protection, with costs split evenly between the federal government, the states and private environmental interests. Colorado agricultural interests were more supportive of environmental objectives, but still preferred less endangered species protection than did Colorado environmental interests.

An analysis of policy attributes found that the dominant attribute in nearly all cases was payment policy (i.e., *Who pays*; see Appendix 6-A, Table 6-A-5). Private environmental interests showed a surprising willingness to support private contribution to the costs of central Platte management programs, and agricultural interests were much more willing to endorse a significant endangered species protection program, if the state cost share was minimized and there was a substantial private contribution. All interest groups were quite receptive to an adaptive management approach that is quite similar to the programs now being pursued by the states and the DOI under the terms of the Cooperative Agreement.

Application of three different sets of bargaining rules all resulted in solutions which called for an adaptive management approach that minimized the reallocation of water, with an equal sharing of the costs between federal, state and private entities. The egalitarian solution, however, suggested that if the agents were more concerned about equity, they should pursue a somewhat more aggressive program of endangered species protection with less of a state contribution to the total cost.

An analysis of the impact of technical knowledge on policy preferences found that well informed people had much stronger environmental preferences compared to those who were less well informed. It was found that much of the disagreement between agricultural and environmental interest groups would cease to exist if both groups had technical beliefs that were similar to those held by well informed individuals. This finding suggests that ecological risk information might have a role in changing public opinion, leading to reduced conflict and perhaps improved resource management. However, there is also a possibility that some respondents knowingly answered technical questions incorrectly, in cases where an incorrect answer supported their strongly held values and policy positions. It is also possible that individuals may reject as biased any new information that did not support such values. Before definitive conclusions can be drawn, further research is needed regarding the effectiveness of ecological-risk education in changing technical beliefs and policy preferences.

6.4 DISCUSSION

Chapter 3 put forward a conceptual approach for the integration of ERA and economic analysis for watershed management (Figure 3-1). In that ideal approach, integration occurs in all stages of assessment. Because economists' involvement began late in the assessment process for the central Platte River floodplain, the process depicted in Chapter 3 was not followed in several respects. That ideal process nonetheless provides a useful framework for evaluating the methods used and degree of ecological-economic integration achieved in this case study.

6.4.1 Assessment planning and problem formulation

The conceptual approach calls for the interrelated steps of assessment planning and problem formulation to be carried out in advance of analysis (Figure 3-1). In this case, a formal planning process that included stakeholders was conducted at the outset of the W-ERA. Planners discussed watershed values and challenges and crafted a very broad management goal – i.e., "to protect, maintain, and where feasible restore biodiversity and ecological processes..." – and a list of eleven management objectives (Section 6.2.1). The W-ERA assessment team then worked to distill those objectives and existing knowledge of the watershed into assessment endpoints, conceptual models and risk hypotheses (Section 6.2.2).

The economic research effort was not yet conceived at this stage, and economists were not involved in this process. The economic study was initiated later, with a coordination meeting that had minimal stakeholder representation and occurred after most of the W-ERA work had already been completed. Therefore, ecological risk assessors did not have the benefit of considering economic concepts, research approaches or management insights, and while economists heard a brief report of the W-ERA approach, they did not benefit from a close collaboration with that effort, nor did they engage a broad range of stakeholder groups in their work. This limited degree of coordination resulted in a divergence of analytic objectives and perspectives. The ecological analysis studied habitat requirements of dozens of riparian-dependent avian species whereas the economic analysis addressed only the needs of endangered species.

6.4.2 Formulating alternatives, and baseline ecological risk assessment

Whereas ERA alone does not necessarily require the formulation of management alternatives, economic analysis usually is concerned with alternatives, so their formulation usually is a condition for integrated study (Figure 3-1). The Platte River W-ERA sought only to characterize baseline risk, i.e., risks that exist now or are likely to occur if no new management is undertaken. The risk models that were developed (i.e., models describing floodplain segment use by sandhill cranes, and meadow or woodland patch use by nesting birds) dealt with a subset of the ecological assessment endpoints. They are potentially applicable to management questions but were not developed with a specific decision context or set of alternatives in mind.

The economic analysis, on the other hand, formulated two sets of management alternatives. Model II focused on finding a compromise solution from among 125 different options (later narrowed to 17 Pareto efficient policies) for floodplain management, especially dealing with instream flow amount and payment. Model I provided a tool, an auction market, for use by stakeholders in deciding who would provide alternative levels of environmental water. The economic analysis thus focused directly on resource management choices that were linked to the dominant issue in the basin, rather than addressing a broader, yet less pragmatically focused, array of baseline ecological risks. Had the economists been part of the W-ERA planning discussions, there would have been an opportunity to discuss these alternatives and thus better harmonize the ecological and economic analyses. Discussion of management alternatives during assessment planning might also have narrowed the scope of the W-ERA, limiting the number of management objectives and risk hypotheses, and sharpening its analytic focus.

6.4.3 Analysis and characterization of alternatives, and comparison of alternatives

The analysis and characterization of management alternatives and the comparison of the alternatives are two closely related steps in the conceptual approach (Figure 3-1). Each management alternative is to be examined in the light of both ecological risks and economic outcomes and, as applicable, other analyses (e.g., health or quality of life). Diagrammatic examples of a variety of approaches to these two steps were given in Figures 3-2, 3-3 and 3-4.

The approach employed in this case study is illustrated in Figure 6-3, which is a modification of Figure 3-3. The likely ecological and economic outcomes of various watershed management policy attributes were described in a survey of preferences, and survey results were used to evaluate specific policies (i.e., attribute combinations).

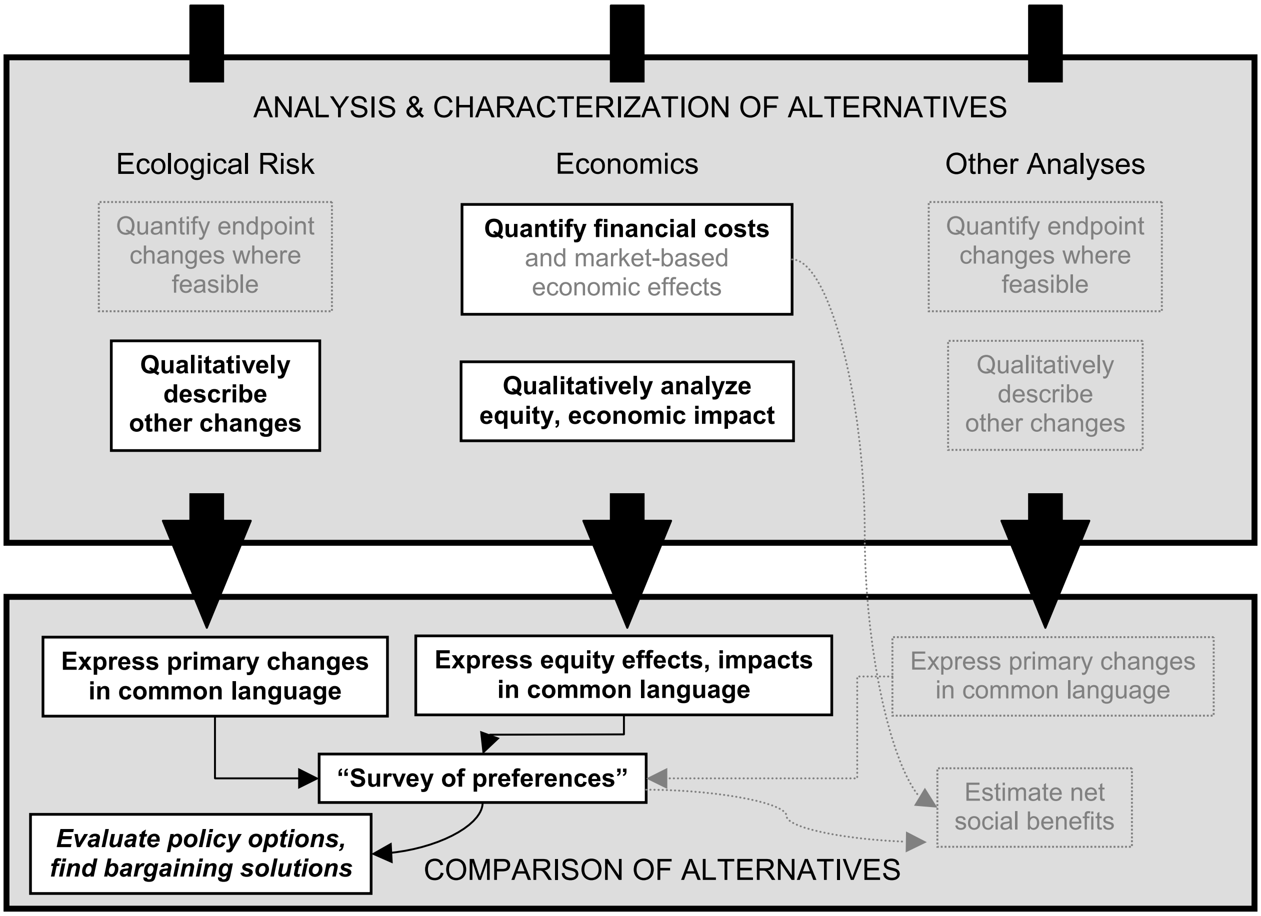

FIGURE 6-3

Techniques used for analysis, characterization and comparison of management alternatives in the central Platte River floodplain, as compared to the example shown in Figure 3-3. White boxes and bold type show features included in this analysis.

The ecological point of departure for the economic study was a determination by USFWS that a given increment of instream flow and restoration of wet meadow acreage are needed to ensure protection of endangered species.[3] This level of provision was described qualitatively in the survey as "meeting the needs" of endangered species; lesser levels of provision were described as placing the species "at risk" (Table 6-8). Annual costs to fund the USFWS program were described in dollar terms. The market-based economic effects of the program (such as the impacts of foregoing water diversion or pumping, or removing land from production) were not estimated or described. However, equity and economic impact concerns were implicit in the wording of policy options that minimized the purchase of water from agriculture or that discussed different cost-sharing options.

In Figure 6-3 the term "survey of preferences" is substituted for "stated preference survey," because the latter usually refers to methods that ask individuals to place a value on specific changes to the environment, whereas in this case the results will not provide estimates of value either directly or indirectly. Analysis of survey results yielded policy-specific estimates of utility for each of several bargaining groups. A subsequent step used the utilitarian, Nash or egalitarian approaches to rank-order the policies. Estimates of the net social benefit of policies could not be derived, in part because market-based economic effects of the policy options were not determined, but also because the survey of preferences did not estimate willingness to pay.

Ecological economics stresses that economic analyses should account for the biophysical constraints that exist in the ecological systems that support all human activity (see Section 2.2.6). The W-ERA for the Platte River did not formulate or evaluate any management alternatives. It is important, therefore, to examine the degree to which the economic models were informed or constrained by information on ecological risks. In general, the economic analysis regarded ecological risk as technical information which could influence the preferences of stakeholders. Ecological risk was constraining only to the extent that stakeholders regarded risk reduction an important objective relative to the trade-offs involved. With this approach no answer is regarded as scientifically correct; all that science does is provide trade-off and preference information to facilitate public decision-making. Model I, the auction model, did several things: (1) it provided a tool for efficiently "negotiating" who will supply a given quantity of water and at what price; (2) it provided a method of estimating the budgetary supply costs associated with different quantities of environmental water; and (3) it provided an indication of the price that stakeholders would pay in the form of welfare and budget costs for using the negotiating efficiencies inherent in a second-price auction instead of a direct negotiation first price approach. Providing these functions required no ecological risk information.

Model II used preference information for policy options that ranged from providing "whatever the USFWS says is needed..." to providing nothing. The Model II bargaining solutions were based on utility and not constrained by conditions ensuring species' survival, beyond respondents' preference for doing so. If respondents preferred policies that were lower-cost or involved less reallocation of water, it is not clear whether they were accepting as valid the biological opinion of the USFWS and voting against full support for maintaining the species, whether they did not believe that water reallocation would be helpful to the species; or were uncertain about key technical relationships and therefore preferred an incremental, try-it-and-see approach. An analysis of the impact of technical information on policy preferences suggested that facts were a very important determinant of policy preferences. Policy preferences changed markedly and the differences between interest groups narrowed substantially if one assumed that with education the less well informed stakeholders would develop preferences similar to those of their better informed colleagues. If this assumption were substantiated, it would raise the possibility that an effective program of educational outreach, carried out in conjunction with a bargaining process, could provide an effective biophysical constraint. However, this study did not investigate the actual effectiveness of education in a situation of longstanding conflict, and therefore it cannot be concluded that the bargaining approach, *per se*, is effectively constrained.

It is possible that the process of adaptive implementation, such as that envisioned by the Cooperative Agreement, would afford constraints ensuring species survival, but much depends on the view one takes of adaptive implementation as a management and political strategy. If it serves as a reliable feedback mechanism, whereby stakeholders' preferences are updated by new information, then biophysical constraints may be effective, even when not explicit in a preference-based model. An adaptive management approach that is politically feasible may reach desired ecological goals at a slower pace than some would prefer, but it may still be the most effective approach if full and immediate implementation is not politically feasible.

6.4.4 Consultation with extended peer community

USEPA's *Ecological Risk Assessment Guidelines* recommend fully involving stakeholders in planning but maintaining strict separation of science from policy in subsequent steps, whereas others have emphasized the limitations of science and the importance of ongoing consultation, throughout the analysis, with an extended peer community (see Sections 2.1.1.5 and 3.3.5). The W-ERA formally established a stakeholder panel for participation in planning, but problem formulation was conducted by a more limited technical team. Near the end of problem formulation, consultations with stakeholders were held and a draft was reviewed, but subsequent changes made by the technical team alienated at least one stakeholder group. The economic analysis was not constrained by a formal requirement for stakeholder involvement and used more limited and informal mechanisms. Lacking their strong involvement, however, it is not yet clear whether the parties to the Cooperative Agreement will make use of the game theory results.

6.4.5 Decisions and adaptive implementation

Even if ecological risk, economics and other information are well integrated and well tuned to the decision context, it is normal for any high-stakes decision to require negotiation after the analyses are completed. The game theory models developed here may be well-suited to the support of an ongoing negotiation because they can respond quickly to changes in negotiating position and suggest new solutions. The approach may also be useful over a longer period of adaptive implementation, in which system modification and feedback result in new learning, and a new set of policy solutions is sought.

Adaptive implementation is important not only for its merit as a management approach but also as an aid to difficult negotiations. When disagreements about the true behavior of the system prevent the parties from agreeing on costly remedies, an adaptive approach can present an attractive compromise in that it holds out the promise of improved knowledge about the system. But care must be taken to distinguish between a policy that is truly adaptive and one that is simply incremental. Walters[66] argues that incrementalism (making small improvements without taking large risks) is not effective as an information-generating strategy. "Such policies result in strongly correlated inputs, and in state variables being correlated with inputs, ... so the effects of each cannot be distinguished." An ideal strategy from an informational standpoint would consist of repetitive sequences moving from one extreme to the other, each of sufficient duration to allow observation of responses of key variables. Managers tend to be risk-averse, however, and under substantial pressure to avoid extremes. An actively adaptive policy, therefore, must somehow establish a balance between learning (via policies designed to maximize probative value) and short-term performance (maintaining the system nearest its status quo).[66]

A key question, therefore, about the value of the Cooperative Agreement as an informative policy is whether the initial increment of 140,000 acre-feet, and evaluation period of 10-13 years, will be sufficient, in light of natural hydrologic variability and the slowness of successional processes, to induce unambiguous changes in key variables such as area of active channel. Since only an unambiguous response would be likely to promote agreement about subsequent actions, the prospects for reducing conflict over the long term through this game theoretic approach are closely tied to adaptive implementation's effectiveness.

6.5 REFERENCES

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APPENDIX 6-A

SUMMARY OF SURVEY RESPONSE INFORMATION USED TO CALCULATE UTILITY OF ENVIRONMENTAL MANAGEMENT POLICY OPTIONS FOR THE CENTRAL PLATTE RIVER FLOODPLAIN

Table 6-8 describes three environmental policy attributes (*Method*, *Cost* and *Who pays*), each having five levels, by which 125 policy options (i.e., 5^3 attribute level combinations) for addressing the central Platte River environmental management problem are defined. Bargaining groups with respect to that environmental problem are determined as a combination of state residency and interest group membership, as defined in Section 6.3.2.3.4 and Table 6-9. Equations 6-3, 6-4 and 6-5 define the methods by which survey response data for several bargaining groups are used to derive each group's utility scores for each policy option. This Appendix summarizes certain information used in the calculation of utility. First, the degree of support for individual policy attribute levels is presented by State (Table 6-A-1) and interest group (Tables 6-A-2, 6-A-3 and 6-A-4). Next, the results of regression analyses conducted to establish the relative weights of the attributes are presented (Table 6-A-5).

TABLE 6-A-1

Degree of support for policy attributes, by state

Policy Attribute and Level[a]	Do Support[b]			Don't Support[c]		
	CO	NE	WY	CO	NE	WY
	Percent of all Respondents					
Method						
a. All Needs, Least Cost	41	24.8	29.2	39.7	49.5	52.7
b. All Needs, Minimum Water	52.6	37.6	37.9	25.8	33.6	35.7
c. Best Possible, No Ag Water	36.7	43.2	45.1	46.7	31.6	34.4
d. Adaptive Management	64.1	63	63.9	21.6	17	17.1
e. Benefit-Cost Approach	26.9	38.7	37.9	47.6	31	36.6
Cost						
a. Invest Nothing	15.7	19.6	23.3	73.1	62.2	59.7
b. Invest $40M, per USFWS	31.8	16.6	19.7	51.7	63.2	63.9
c. Invest $10M, 25% of Need	36.2	39	30.6	39.4	37.6	43.8
d. Invest $20M, 50% of Need	33.6	22.4	18.9	42.4	51.5	54.3
e. Invest $30M, 75% of Need	23.8	13.2	11.7	48.4	57.6	60.3
Who pays						
a. All Costs Paid by Feds	32.2	34.9	33	57	48.7	51.7
b. Feds 50%, Private 50%	39.9	39.6	49.2	44.5	39.8	35
c. Feds 50%, States 50% Equal	27.8	26.2	17.5	53.9	54.7	64.3
d. Feds 50%, States 50% Prop.	43.5	29.4	34.4	37.9	46.6	45.1
e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use	61.6	51.3	53.2	25	29.6	31.1

[a]A full description of each policy attribute and level is found in Table 6-8.
[b]Includes responses of "strongly support" and "support."
[c]Includes responses of "strongly oppose" and "oppose."

TABLE 6-A-2

Degree of support for policy attribute levels in Colorado, by interest group

Policy Attribute and Level[a]	Do Support[b]		No Opinion		Don't Support[c]	
	Ag	Envl.	Ag	Envl.	Ag	Envl.
	Percent of Classified Respondents					
Method						
a. All Needs, Least Cost	26.1	56.7	4.3	22.7	69.6	20.6
b. All Needs, Minimum Water	60.9	69.3	13	22.1	26.1	8.6
c. Best Possible, No Ag Water	73.9	22.9	8.7	15.7	17.4	61.4
d. Adaptive Management	87	55.3	8.7	12.8	4.3	31.9
e. Benefit-Cost Approach	43.5	17.7	26.1	27	30.4	55.3
Cost						
a. Invest Nothing	49.9	4.4	4.5	7.3	54.5	88.3
b. Invest $40M, per USFWS	0	58	9.1	13.8	90.9	28.3
c. Invest $10M, 25% of Need	56.5	30.8	8.7	27.1	34.8	42.1
d. Invest $20M, 50% of Need	18.2	40	9.1	24.4	72.7	35.6
e. Invest $30M, 75% of Need	4.5	38.1	13.6	28.4	81.8	33.6
Who pays						
a. All Costs Paid by Feds	29.4	37.7	4.3	9.4	66.5	52.9
b. Feds 50%, Private 50%	58.8	39.7	8.7	17.6	39.1	42.6
c. Feds 50%, States 50% Equal	5.9	43.8	0	22.6	87	33.6
d. Feds 50%, States 50% Prop.	5.9	58.7	0	23.2	87	18.1
e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use	47.1	18.4	4.2	18.4	45.8	13.2

[a]A full description of each policy attribute and level is found in Table 6-8.
[b]Includes responses of "strongly support" and "support."
[c]Includes responses of "strongly oppose" and "oppose."

TABLE 6-A-3

Degree of support for policy attribute levels in Nebraska, by interest group

Policy Attribute and Level[a]	Do Support[b]		No Opinion		Don't Support[c]	
	Ag	Envl.	Ag	Envl.	Ag	Envl.
	Percent of Classified Respondents					
Method						
a. All Needs, Least Cost	19.6	38.8	14.7	24.7	65.7	36.5
b. All Needs, Minimum Water	35.6	52.9	19.8	28.2	44.6	18.8
c. Best Possible, No Ag Water	57.3	27.1	17.5	21.2	25.2	51.8
d. Adaptive Management	69.9	56.5	12.6	21.2	17.5	22.4
e. Benefit-Cost Approach	47.1	22.4	29.4	23.5	23.5	54.1
Cost						
a. Invest Nothing	35.6	7.1	16.8	8.2	47.5	84.7
b. Invest $40M, per USFWS	9.8	32.1	13.7	23.8	76.5	44
c. Invest $10M, 25% of Need	35	44.4	16	18.5	49	37
d. Invest $20M, 50% of Need	21.6	30.5	17.6	22	60.8	47.6
e. Invest $30M, 75% of Need	5.9	33.3	18.8	25	75.2	41.7
Who pays						
a. All Costs Paid by Feds	38	37.3	12	20.5	50	42.2
b. Feds 50%, Private 50%	50	36.1	15	20.5	35	42.4
c. Feds 50%, States 50% Equal	19	34.5	15	20.2	66	45.2
d. Feds 50%, States 50% Prop.	24.8	44.6	15.8	24.1	59.4	31.3
e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use	41	54.8	16	23.8	43	21.4

[a]A full description of each policy attribute and level is found in Table 6-8.
[b]Includes responses of "strongly support" and "support."
[c]Includes responses of "strongly oppose" and "oppose."

TABLE 6-A-4

Degree of support for policy attribute levels in Wyoming, by interest group

Policy Attribute and Level[a]	Do Support[b]		No Opinion		Don't Support[c]	
	Ag	Envl.	Ag	Envl.	Ag	Envl.
	Percent of Classified Respondents					
Method						
a. All Needs, Least Cost	13.2	61.1	11.3	13.9	75.5	25
b. All Needs, Minimum Water	34.6	52.8	13.5	28.7	51.9	18.5
c. Best Possible, No Ag Water	71.2	18.7	15.4	15.9	13.5	65.4
d. Adaptive Management	83.3	48.1	9.3	19.4	7.4	32.4
e. Benefit-Cost Approach	50.9	23.6	22.6	19.8	26.4	56.6
Cost						
a. Invest Nothing	42.3	6.5	19.2	5.6	38.5	88
b. Invest $40M, per USFWS	3.8	50	13.5	10.2	86.7	39.8
c. Invest $10M, 25% of Need	37.7	27.8	18.9	28.7	43.4	43.5
d. Invest $20M, 50% of Need	13.7	27.5	19.6	26.6	66.7	45.9
e. Invest $30M, 75% of Need	0	25.9	19.6	28.7	80.4	45.4
Who pays						
a. All Costs Paid by Feds	37	34.9	7.4	12.8	55.6	52.3
b. Feds 50%, Private 50%	60.4	41.1	7.5	14	32.1	44.9
c. Feds 50%, States 50% Equal	5.9	36.2	5.9	15.2	88.2	48.6
d. Feds 50%, States 50% Prop.	13.5	61.1	15.4	14.8	71.2	24.1
e. Feds 1/3, Pvt.1/3, States 1/3 Proportional to Use	47.1	57.5	3.9	12.3	49	30.2

[a]A full description of each policy attribute and level is found in Table 6-8.
[b]Includes responses of "strongly support" and "support."
[c]Includes responses of "strongly oppose" and "oppose."

TABLE 6-A-5

Policy attribute weights by bargaining group[a]

Interest Group	Intercept	*Method*, M	*Cost*, C	*Who pays*, P
Colorado, State, N = 994				
Reg. Coefficients, B	0.772	0.211	0.119	0.413
Standard Error		0.021	0.020	0.020
Normalized Weights, W		0.28	0.16	0.56
Colorado Agricultural, N = 154				
Reg. Coefficients, B	0.855	0.068	0.382	0.242
Standard Error		0.070	0.081	0.087
Normalized Weights, W		0.10	0.55	0.35
Colorado Environmental, N = 840				
Reg. Coefficients, B	0.873	0.191	0.192	0.321
Standard Error		0.030	0.030	0.030
Normalized Weights, W		0.27	0.27	0.46
Nebraska State, N = 1,179				
Reg. Coefficients, B	0.900	0.093	0.204	0.387
Standard Error		0.017	0.018	0.017
Normalized Weights, W		0.14	0.30	0.57
Nebraska Agricultural, N = 674				
Reg. Coefficients, B	0.628	0.056	0.198	0.524
Standard Error		0.031	0.036	0.035
Normalized Weights, W		0.07	0.25	0.67
Nebraska Environmental, N = 505				
Reg. Coefficients, B	1.729	0.055	0.026	0.332
Standard Error		0.043	0.043	0.043
Normalized Weights, W		0.13	0.06	0.81
Wyoming State, N = 999				
Reg. Coefficients, B	0.663	0.129	0.198	0.420
Standard Error		0.018	0.019	0.018
Normalized Weights, W		0.17	0.26	0.56
Wyoming Agricultural, N = 646				
Reg. Coefficients, B	0.840	0.078	0.177	0.396
Standard Error		0.043	0.054	0.050
Normalized Weights, W		0.12	0.27	0.61
Wyoming Environmental, N = 353				
Reg. Coefficients, B	1.035	0.154	0.117	0.370
Standard Error		0.031	0.032	0.030
Normalized Weights, W		0.24	0.18	0.58

[a] See Equations 6-4 and 6-5 for explanation of variables and attribute weights

7. CONCLUSIONS

This document has introduced fundamental concepts and methods in ecological risk assessment (ERA) and economic analysis of environmental problems, especially as applied to watersheds (see Chapters 1 and 2), and it has developed a conceptual approach for their integration in watershed management (see Chapter 3, and especially Figure 3-1). It has described and evaluated case studies of three U.S. watersheds in which watershed ERA (W-ERA) was conducted, followed by economic analysis that utilized the W-ERA findings (Chapters 4-6). This closing chapter draws general conclusions from this research effort. For the most part, it leaves aside issues that are particular either to ERA itself or to economic analysis and focuses on the problem of their integration.

These conclusions do not constitute a comprehensive list of recommendations for integrating ERA and economic analysis. The conceptual approach for integration presented in Chapter 3 is more complete in that regard. Rather, they are a set of important observations drawn from an overview of these three case studies. The conclusions provide further insight on certain topics raised by the conceptual approach, but additional studies are still needed to explore that approach more fully.

7.1 ACHIEVING ECOLOGICAL-ECONOMIC INTEGRATION REQUIRES A COHERENT STRATEGY

The central conclusion arising from evaluation of the case studies is that watershed problems should be approached with a coherent strategy for assessment and management. If decision-makers need to consider both ecological risks and economic factors (and perhaps other factors), a strategy that guides their integration is necessary. The conceptual approach described in Chapter 3 provides such a strategy. The approach is based on the U.S. Environmental

Protection Agency (USEPA) *Framework for Ecological Risk Assessment*,[1,2] and it modifies or augments that framework as needed to accommodate economic analysis, and to address a broader management context. Its elements are similar to those of other frameworks that have been used in environmental management (see Table 3-4 and Appendix 3-A). Although this document presents the conceptual approach before the case studies, to serve as a guide to their evaluation, it was developed following their completion and should be considered the main outcome of this body of investigation.

The case studies help illustrate the need for the conceptual approach. The W-ERA studies were not undertaken with economic integration as a goal. The economic studies did have such a goal, but used only a limited set of guiding principles; i.e., each economic analysis was to address the same system, problems and ecological assessment endpoints analyzed by the W-ERA, and it was to be relevant to decision-making. The approaches used were novel and the results are potentially useful, but in each case their usefulness could have been improved by a more comprehensive approach, as is detailed in the following sections. For example, the lack of an interdisciplinary assessment planning and problem formulation process contributed in one case to divergent views of goals and endpoints. In two cases (Clinch and Platte), management alternatives were formulated for economic analysis, but the likely ecological effects of those alternatives were not quantitatively assessed, limiting the scope of the conclusions. Also, in two cases (Darby and Clinch) the economic analysis tools chosen were not clearly aligned to the relevant decision context; that is, it was not shown that they were developed with a set of decisions and decision-makers in mind. Use of the conceptual approach for integration theoretically could have helped avoid these limitations.

It is unlikely however, that ideal conditions will often exist in which ecologists, economists, other specialists and stakeholders can make a clean start to define a problem together, using an inclusive, analytic process. More likely is the kind of situation described in these case studies, where some baseline of study and stakeholder involvement has been established and an effort is made later to inject additional elements. Although it may be infeasible to restart the entire process, it is nonetheless advisable to revisit key portions of the early steps of assessment, with stakeholder involvement, so as to harmonize management objectives, decision context, management alternatives and assessment endpoints to the extent possible. It is also important to use the guiding considerations presented in Section 3.2 (see Table 3-2) to identify ways to make ongoing efforts more integrated in character.

7.2 INTEGRATION REQUIRES ASSESSMENT PLANNING AND PROBLEM FORMULATION TO BE INTERDISCIPLINARY

The conceptual approach emphasizes the need for ecologists and economists (and other specialists as required) to participate together in the steps of assessment planning and problem formulation. The fact that the ERA and economic analysis were done sequentially in these case studies, rather than in a more integrated fashion, limited their value for management. In the Big Darby Creek watershed of Ohio, a team of ecologists and economists from Miami University built upon a W-ERA that had been initiated several years earlier by USEPA, Ohio EPA and a number of other partners.[3-5] Economic analysis in the Clinch Valley of Virginia and Tennessee was by an interdisciplinary team headed by the University of Tennessee-Knoxville (UT-K) and used the results of a W-ERA previously conducted by USEPA, U.S. Fish and Wildlife Service (USFWS), Tennessee Valley Authority and other partners.[6-8] In the central reach of the Platte River, a study by economists from the University of Nebraska-Lincoln (UN-L) built upon the foundations of a W-ERA that had been initiated by USEPA, UN-L, USFWS, and U.S. Geological Survey, with participation by a host of local stakeholder groups.[9-11]

In the W-ERA efforts, planning and problem formulation were systematic and painstaking, but economists were not involved. When the economic studies were initiated, informational meetings were held with members of the W-ERA teams, but these did not reopen fundamental questions about the management problems, so views about management goals and objectives were not necessarily the same.

The lack of a common view was most pronounced in the Platte River case study. The W-ERA team viewed the vegetative diversity and dynamic character of the braided-river-channel landscape mosaic as an endpoint in itself, as well as the diversity of fauna using its various habitats. The economic team focused more narrowly on current efforts among the three Platte River states and the federal government to reach agreement on provisions to meet the needs of three endangered species, the interior least tern (*Sterna antillarum athalassos*), the piping plover (*Charadrius melo*dus) and the whooping crane (*Grus americana*). The W-ERA analyzed conditions affecting the use of river segments by sandhill cranes (*Grus canadensis*), whose needs overlap substantially with the endangered species', but they also analyzed the effects of landscape patch size on grassland and woodland breeding birds, whose needs are less relevant to, and in some cases conflict with, those of the endangered species. In the other two case studies there was not a significant divergence of views; however, the lack of a joint assessment planning exercise may have contributed to the failure to identify a decision context for the economic assessment, as well as certain other weaknesses discussed below.

7.3 RESEARCH IS NEEDED ON THE DEVELOPMENT AND USE OF INTEGRATED CONCEPTUAL MODELS

A conceptual model is a graphical depiction, typically a box-and-arrow diagram, of the hypothesized relationships between human activities, ecological stressors, and ecological assessment endpoints (refer to Section 2.1.1.2 for explanation, and Figure 5-3 for an example). According to the conceptual approach for integration, an interdisciplinary problem formulation process should include the development of extended conceptual models (see Section 3.3.2). In extended models, risk hypotheses show how sources and stressors affect economic endpoints, or services, as well as ecological assessment endpoints. An extended model also includes risk management hypotheses, which we have defined as explanations of how management alternatives are expected to affect sources, exposures, effects and services. Their development should involve environmental program managers, if the management actions are in the form of programs or policies, and environmental engineers or restoration specialists, if the actions involve structural changes to ecosystems. Their development also requires the involvement of land owners and other stakeholders whose active cooperation may be instrumental in solving the environmental problem. Extended models were not developed in these case studies, and at present we are not aware of examples of the use of these extended models in a risk assessment context. The National Center for Environmental Assessment of USEPA's Office of Research and Development is presently initiating work to gain experience with their development and use.

7.4 CLEARLY FORMULATED MANAGEMENT ALTERNATIVES FACILITATE INTEGRATED ANALYSIS

Describing management alternatives is an important way to frame the integration problem. Any given alternative will entail a unique set, or bundle, of ecological, economic and other kinds of changes. Some of those changes may be judged beneficial and some detrimental, some to a larger and others to a lesser degree. The heart of the integration problem is to somehow evaluate the signs and magnitudes of all these changes collectively, on a common scale, to determine if one alternative can be clearly preferred over another.

In the Big Darby watershed, three possible land use scenarios (low-density ranchettes, low-density cluster, and maintaining agriculture) and a most-likely base case (high density residential) were described in some detail, and their respective ecological, economic and quality-of-life impacts were determined. Using the contingent valuation method (CVM), the researchers were able to jointly value the different impacts. Although each respondent was posed only one of the three possible choices, mean willingness to pay (WTP) serves as a kind of referendum on these three alternatives.

In the Clinch Valley study, two hypothetical policies for establishing voluntary agriculture-free riparian zones (i.e., a narrow zone and a wider zone), compensated by property or income tax revenues, were employed in a conjoint survey. The choice sets included in the survey were generated as random combinations of these policies and other attributes describing potential ecological outcomes and individual payments, and therefore the choices did not correspond to specific policy scenarios. However, the resulting choice model could be used to generate a mean WTP for obtaining any policy scenario that could be described from those attributes (as compared to the status quo), and such a value would have an interpretation similar to the Big Darby result. As mentioned above, however, the expected ecological outcomes of such a policy were not estimated.

The Platte study differed in that the problem of determining a preferred policy was viewed not as one of determining mean WTP but rather as determining what policy certain competing factions were most likely to find mutually acceptable. Like the other two, it elicited responses to preference questions that combined ecological and economic dimensions, but unlike them it used this information to model a negotiation process. Using principles from game theory (the study of interacting decision-makers),[12] the model analyzed 125 hypothetical policies for meeting endangered species needs, where a given policy described the method of meeting those needs, its cost and who would pay. As in the Clinch case study, the expected ecological outcomes of the policies were not analyzed. Since management alternatives are important for economic analysis and for decision-making, their formulation should receive careful attention from all parties involved in the assessment, and their ecological outcomes should be estimated.

7.5 CAREFUL EFFORT IS REQUIRED TO RELATE ECOLOGICAL ENDPOINTS TO ECONOMIC VALUE

An important step in the problem-formulation phase of ERA is the selection of ecological assessment endpoints. Assessment endpoints are chosen that are considered ecologically relevant, susceptible to the stressors of concern and relevant to the environmental management objectives (see Section 2.1.1.2). The likelihood of adverse effects on these endpoints is described in the risk-characterization phase (see Section 2.1.1.4). The challenge of ERA-economic integration includes determining economic value (see Section 2.2.2) associated with those changes as well as characterizing other linkages between the ecological system, management actions and economic value (see Section 3.3.2).

In these case studies, endpoints chosen for ERA because of their ecological importance sometimes posed a challenge for economic analysis. Whereas the freshwater mussel faunas of the Big Darby and Clinch systems are considered ecologically significant, members of the general public who are unaware of their diversity and threatened status may be unconcerned about their survival. To counter this problem, in the Clinch study the survey text mentioned mussels ten times in its brief introductory paragraphs, explaining their unusual degree of diversity in the Clinch Valley, their usefulness as an indicator of water quality, and their sensitivity to pollutants and susceptibility to crushing by the hooves of cattle, before posing the choice sets. In the central reach of the Platte River, where management concerns have centered on endangered or conspicuous migratory waterfowl, the landscape-ecological viewpoint employed in the W-ERA treated landscape diversity, and several less conspicuous bird species, as additional endpoints. These endpoints did not factor in the economic (i.e., game theoretic) analysis, but if efforts had been made to value these endpoints, similar problems would have been faced.

Another complication occurred when the measurement methods that were used to express the ecological endpoints, or were a surrogate for the endpoints, were not readily understandable to the public. For example, even if the public considers a diverse stream fauna to be important, they may have difficulty determining what they would be willing to give up in order to obtain, e.g., a 3-point or 10-point improvement in a multimetric ecological index. Since these indices may be composites of ten or so individual measures, it is impossible to make a scientifically precise statement about the meaning of any such change. Yet because these indices are becoming widely relied upon to indicate the presence or absence of biological impairment (Section 2.3.1 and Appendix 2-B), they are likely to be a critical part of the available knowledge base about ecological risk in a watershed, and ways must be found to adequately communicate their meaning if individuals are to determine how such changes affect their welfare. In the Big Darby CVM study, the index of biotic integrity (IBI, a fish assemblage indicator) and invertebrate community index (ICI, a stream-bottom community indicator) were used as risk assessment endpoints. CVM survey respondents were shown a table (see Table 4-1) in which each of the four land use scenarios was rated from “low” to “high” for each of four stressors (nutrients, sediments, toxins and flow pattern) and were told that a “high” level posed a “risk to stream integrity.” In the Clinch Valley study, where the study of risks relied heavily on IBI, respondents were presented with choice sets in which one of six attributes of the choice was “aquatic life,” and the possible levels were “full recovery,” “partial recovery” or “continued decline” (see Tables 5-3 and 5-4). The supporting text (see Appendix 5-A) explained that “partial recovery” meant “some improvement” in the Clinch River but not in its tributaries, whereas “full recovery” meant “improvement” in both the Clinch River and its tributaries. Both surveys avoided direct presentation of the indices, using instead qualitative description. While the descriptors for the Darby study could be related back to the results of scenario impact analysis, those for the Clinch were not as easily related to a given physical change.

Where environmental management may have particular objectives—for example, the protection of water quality and stream biological integrity—the results of management actions can affect additional endpoints as well. Therefore, the ecological information set needed for economic analysis may be broader than that envisioned in the ERA (if problem formulation for the ERA did not include consideration of management alternatives). In the Clinch Valley, for example, management actions examined in the economic study included hypothetical policies to compensate farmers for voluntarily restricting agriculture from a riparian buffer area. Besides improvements in diversity of native mussels and fish, which were the ERA endpoints, the economists expected that such policies would improve sport fishing and enhance the presence of songbirds, which were not included in the ERA. Consideration of songbirds turned out to be unimportant in this case, since respondents did not appear to value them significantly (see Table 5-8), but sport fishing was important. Full analysis of the economic benefits of these policies therefore would have required analysis of sport fish response. Other potential economic benefits of riparian zone restoration may result from enhanced nonavian wildlife habitat, reduced nutrient export, increased sequestration of carbon and improved value of river-corridor recreation such as canoeing. Had an attempt been made to capture these values as well, additional ecological and economic endpoints only tangentially related to the original management goal would have been required.

Estimating the benefits of a given change in the ecological condition of water bodies requires better procedures. In 1986, Mitchell and Carson[13] reported on a national survey of U.S. households to quantify water quality-related WTP. They used a "water quality ladder" that established progressively increasing use levels (i.e., boating, game fishing, swimming, drinking) for surface waters. These levels equated to points on a cardinal scale determined as a combined index of five conventional water quality parameters: fecal coliforms, dissolved oxygen, biological oxygen demand, turbidity and acidity (pH). Thus, the benefit of any change in those parameters could be associated with WTP, but the index reflected only a narrow set of pollutants and did not include any direct measurements of stream biological communities.

Since then, substantial progress has been made in the development of state programs for biological monitoring and the use of indices such as IBI and ICI in water quality standards (WQS). These programs have not required a detailed understanding on the public's part of the measures that underlie the indices or a feel for their numerical scales. Work must be done to expand the scientific basis of the water quality ladder to include a broader set of ecological measures, or in some cases to replace exposure (pollutant) measures with response (biological) measures. In addition, the uses, or rungs, that were originally examined need to be expanded to better reflect the full variety of uses that have been designated in state WQS programs (personal communication with John Powers, USEPA Office of Water) as well as other levels of quality to which the public may attribute value. For example, respondents in the Big Darby and Clinch Valley studies probably recognized freshwater mussels as valued components of those aquatic communities, yet a level of water quality sufficient to support game fish (the highest rung of the ladder) may not be sufficient to promote "full recovery" of mussels.

Thus, informed decisions (i.e., ones where decision-makers understand the inherent trade-offs) require techniques that link the kinds of indices ecologists currently measure to values held by the public. Part of the challenge, therefore, is translating indicators into common language.[14] By the same token, ecological measures may require adjustment. For example, if the public values the response of the instream biological community to stream corridor restoration, then ecological measures of the efficacy of such projects should not be limited to modeled changes in water quality parameters. Similarly, if sport fishing and bird watching are among the values the public places on such projects, then measuring aquatic community integrity alone is not sufficient. Further, since ecological measurements are highly variable, and model predictions highly uncertain, research needs to include methods to enable the public to understand and account for ecological uncertainty in their preferences.

7.6 THE APPROPRIATE TOOLS FOR ANALYSIS AND COMPARISON OF ALTERNATIVES DEPEND ON THE DECISION CONTEXT

To weigh management alternatives, analysts should select comparison methods that fit the decision context. If decision criteria are constrained by statute or regulation, then the comparison procedure must include any required information and be capable of segregating any precluded information. For example, regulatory impact analyses conducted by USEPA (see Section 2.3.2) may require an analysis in which all costs and benefits are monetized to the greatest extent feasible. By contrast, U.S. Army Corps of Engineers project evaluation procedures maintain separate accounts for changes in the national output of goods and services (expressed in monetary units) and changes in the net quantity and/or quality of desired ecosystem resources (expressed in physical units).[15] These decision contexts imply particular comparison procedures, whereas in other contexts procedures can be subjective or ad hoc. Other important differences in context may be as follows:

- *one entity* has clear authority to decide vs. *many parties* will negotiate
- *one decision* will be made affecting a large area vs. *many small decisions* will be made, each affecting only one land parcel, stream segment or political jurisdiction
- decision-makers expect to reach a *decision point* once analysts have presented all information vs. decision-makers expect to examine data, construct alternatives and engage in an active *decision process*.

To ensure successful integration researchers need to categorize environmental management situations based on the decision context and evaluate the full complement of comparison procedures available, in order to identify compatibilities between context and procedure.

All three of the case studies surveyed watershed residents, and in some cases other members of the public, and used information about respondents' preferences to produce tools that integrated ecological and economic factors. These tools are potentially useful in decision-making and management. In the Big Darby case, the Miami team developed a broadly framed, contingent valuation method (CVM) approach for comparing economic, ecological and quality-of-life outcomes among four alternative futures. Preferences were expressed via a WTP measure that is consistent with standard economic theory and therefore useful in a variety of decision contexts.

The UT-K study developed a choice model to measure valley residents' preferences regarding hypothetical riparian management policies in the upper Clinch and Powell Rivers. Since the model's parameters correspond to a set of attributes of the choice, the model is sufficiently flexible that it could be used (in conjunction with expert judgment) to refine the design of an actual policy so as to maximize its value to Clinch Valley residents.

In the central reach of the Platte River, the UN-L economists modeled the utility of a large set of potential policies from the perspectives of different groups. They also investigated an auction method whereby the states of Colorado, Wyoming and Nebraska could more readily agree on water price and supplier.

The tool developed by each team has potential for application to management problems in the watershed studied, and the methods involved could be adapted to other environmental settings. The CVM approach taken in the Darby presented concrete choices in readily understandable terms. Although the choices between development scenarios were hypothetical, the visual impact of photographic examples of each kind of development as it occurs in the watershed made the choice very realistic. On the other hand, the attribute-driven models developed as part of the Clinch and Platte analyses afforded greater analytic flexibility, which could have substantial value in later phases of management. Since negotiation between affected parties can play an important role in decision-making, an analytic approach that can respond quickly to changes in design (i.e., without the requirement of a new survey) may be very useful. If an incremental or adaptive (learn-as-you-go) implementation approach is to be used, a flexible model would be preferred that could be revised and rerun using newly acquired information about the effectiveness of the management approach. However, if the public is becoming more informed in the process, then a new survey may be required in any case.

The game theoretic approach employed in the Platte River case was carefully selected to fit a specific decision context, i.e., a tristate negotiation to meet Endangered Species Act requirements. On the other hand, the tools developed for the Darby and Clinch watersheds provided information about regional development preferences but were not directed at a particular set of decision-makers. Thus, although the latter tools are potentially useful, it is less clear that they are the best tools for management of those watersheds.

7.7 RESEARCH IS NEEDED ON TRANSFERRING THE VALUE OF ECOLOGICAL ENDPOINT CHANGES

Environmental management problems tend to be highly unique, complicating the direct transfer of economic findings from one situation to another. A given watershed under study is likely to differ in some key characteristic from another where a similar problem may already have been studied. The novel methods developed in each of these case studies undoubtedly could be adapted for use in other systems. Given the expense and time of conducting surveys, however, analysts need to understand whether there are dimensions of value that are less variable across systems. The Big Darby results suggest that one might be able to improve the comparability of WTP estimates by using the numerical IBI change and area affected (or perhaps stream miles affected) as normalizing factors. Work in that case is still ongoing to determine if ecological value can be estimated as a fraction of WTP. The Clinch Valley case study decomposed WTP according to a set of attributes, determining part-worths for each. We might hypothesize that such a partial value, if normalized for magnitude and extent of stream improvement, would be less variable across situations than would a more bundled estimate. These assumptions require validation, however.

7.8 THE ROLE OF ECOLOGICAL RISK INFORMATION IN THE MEASUREMENT OF PREFERENCES REQUIRES FURTHER RESEARCH

Individuals' preferences about uncertain outcomes reflect their expectations about those outcomes, and expectations depend on beliefs.[16] When individuals know little about an environmental management problem, the information provided in a survey will have an important influence on the construction of beliefs and the statement of preferences.[17] The purpose of ERA is to develop accurate information about the nature, magnitude and certainty of adverse effects to ecological resources, given present circumstances and sometimes under different prospective management regimes. The challenge of integration therefore goes beyond determining how to associate preferences with risk outcomes; it includes determining the appropriate use of risk information to inform (or even construct) preferences.

The treatment of information and belief used in the Clinch Valley study was the most conventional, in that the mail-out questionnaire included some introductory and explanatory text (including discussion about mussels, as pointed out earlier) to help respondents understand the questions, and it asked questions about respondent age, income, education, environmental beliefs and affiliations to help characterize the respondent population and determine the factors that underlie preference. The Platte River study similarly employed a mail-out survey with an informative preamble and demographic questions, but it took the additional step of asking respondents' agreement or disagreement with a series of statements about the environmental management problem. These were intended to determine not only attitudes but knowledge, since the statements were considered to have known, correct answers,[a] and responses were used to

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[a] In reality there was some ambiguity about this distinction, since not all of the answers could be clearly established by documentation.

score respondents' knowledge level (see Section 6.3.2.1.2). This information was then used to speculate about the potential effects of better information on negotiation outcomes. By contrast, the Big Darby survey used an in-person presentation approach, with a detailed script and a computer-based slide show including many photographs, to clearly illustrate each of the development scenarios and their anticipated outcomes. Risk assessors have often recognized risk communication as an important field for research and development of practical techniques. The differing approaches used in these surveys highlight the importance of defining best practices and exploring novel techniques for risk communication in survey design and in other stages of decision-making.

7.9 FINAL WORD

Because watershed boundaries often encompass areas that are ecologically and socially complex, assessment and management of watershed problems can be complex as well. Processes to support watershed decision-making need to be flexible and adaptable to a given context, and multidisciplinary analyses are often required. Differences in methodology between the disciplines, especially between the natural and social sciences, can complicate the decision-making task, but as these case studies have shown they also provide fertile ground for the development of unique approaches. The conceptual approach for integration of ERA and economic analysis presented in this document offers a set of principles and procedures that can help ensure that analyses are constructively focused and mutually supportive. It also offers a coherent framework within which other novel, analytical approaches should be explored.

7.10 REFERENCES

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