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ECOLOGICAL RISK ASSESSMENT ISSUE PAPERS

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EPA Work Group Chairs, Authors, and Peer Reviewers

Issue Paper on Ecological Significance

EPA Work Group

John Gentile, Office of Research and Development (Chair)
Pat Cirone, Region 10
Anne Fairbrother, Office of Research and Development
Harvey Simon, Region 2
Daniel Vallero, Office of Research and Development

Issue Paper Authors

Mark Harwell, University of Miami
Bryan Norton, Georgia Institute of Technology
William Cooper, Michigan State University
John Gentile, Office of Research and Development

Issue Paper Peer Reviewers

Kent Thornton, FTN Associates (Lead Reviewer)
Robert Bachman, Maryland Department of Natural Resources
Thomas O'Connor, National Oceanic and Atmospheric Administration

Issue Paper on Conceptual Model Development

EPA Work Group

Donald Rodier, Office of Pollution Prevention and Toxics (Chair)
Craig Barber, Office of Research and Development
Pat Cirone, Region 10
Anne Fairbrother, Office of Research and Development

Issue Paper Authors

Lawrence Barnthouse, Oak Ridge National Laboratory
Joel Brown, University of Illinois/Chicago

Issue Paper Peer Reviewers

Gregory Biddinger, Exxon Corporation (Lead Reviewer)
Ronald Kendall, Clemson University
Robert O'Neill, Oak Ridge National Laboratory

Issue Paper on Characterization of Exposure

EPA Work Group

Sue Norton, Office of Research and Development (Chair)
Richard Batiuk, Region 3
Sandra Bird, Office of Research and Development
Russell Kinerson, Jr., Office of Water
John Paul, Office of Research and Development
Eric Preston, Office of Research and Development

Issue Paper Authors

Glen Suter, Oak Ridge National Laboratory
James Gillette, Cornell University
Sue Norton, Office of Research and Development

Issue Paper Peer Reviewers

William Adams, ABC Laboratories (Lead Reviewer)
Lawrence Kapustka, Ecological Planning and Toxicology, Inc.
Frederic Wagner, Utah State University

Issue Paper on Effects Characterization

EPA Work Group

Anthony Maciorowski, Office of Pesticide Programs (Chair)
Richard Bennett, Office of Research and Development
Steven Bradbury, Office of Research and Development
Jasper Garner, Office of Research and Development
James Lazorchak, Office of Research and Development
H. Ronald Preston, Region 3

Issue Paper Authors

Patrick Sheehan, McLaren/Hart
Orie Loucks, Miami University (Ohio)

Issue Paper Peer Reviewers

Jeffrey Giddings, Springborn Laboratories (Lead Reviewer)
Nelson Beyer, National Biological Survey
Wayne Landis, Western Washington State University

Issue Paper on Biological Stressors

EPA Work Group

Robert Frederick, Office of Research and Development (Co-Chair)
David Yount, Office of Research and Development (Co-Chair)
Fred Betz, Office of Pesticide Programs
Anne Fairbrother, Office of Research and Development
Sharlene Matten, Office of Prevention, Pesticides, and Toxic Substances
Suzanne Marcy, Office of Water

Issue Paper Authors

Daniel Simberloff, Florida State University
Martin Alexander, Cornell University

Issue Paper Peer Reviewers

James Drake, University of Tennessee (Lead Reviewer)
Richard Orr, U.S. Department of Agriculture
James Thorp, University of Louisville

Issue Paper on Ecological Recovery

EPA Work Group

Michael Brody, Office of Policy, Planning, and Evaluation (Chair)
Naomi Detenbeck, Office of Research and Development
Suzanne Marcy, Office of Water
David Yount, Office of Research and Development

Issue Paper Authors

Stuart Fisher, Arizona State University
Robert Woodmansee, Colorado State University

Issue Paper Peer Reviewers

James Karr, University of Washington (Lead Reviewer)
Patrick Brezonik, University of Minnesota
Randall Wentsel, U.S. Senate Committee on Environment and Public Works

Issue Paper on Uncertainty in Ecological Risk Assessment

EPA Work Group

David Mauriello, Office of Pollution Prevention and Toxics (Chair)
Craig Barber, Office of Research and Development
Russell Kinerson, Office of Water
Lorenz Rhomberg, Office of Research and Development
Kevin Summers, Office of Research and Development

Issue Paper Authors

Eric Smith, Virginia Polytechnic Institute and State University
H. Shugart, University of Virginia

Issue Paper Peer Reviewers

A. Frederick Holland, South Carolina Marine Research Resources Institute (Lead Reviewer)
Lev Ginsburg, State University of New York/Stoney Brook
Kenneth Rose, Oak Ridge National Laboratory

Issue Paper on Risk Integration Methods

EPA Work Group

Lawrence Burns, Office of Research and Development (Chair)
Russell Kinerson, Office of Water
Richard Lee, Office of Pesticide Programs
Peter Principe, Office of Research and Development
Paul Zubkoff, Office of Pesticide Programs

Issue Paper Authors

Richard Wiegert, University of Georgia
Steven Bartell, Senes Oak Ridge, Inc.

Issue Paper Peer Reviewers

John Bascietto, Department of Energy (Lead Reviewer)
John Geisy, Michigan State University
Peter Van Voris, Battelle Pacific Northwest Laboratories

Issue Paper on Ascertaining Public Values Affecting Ecological Risk Assessment

EPA Work Group

John Gentile, Office of Research and Development (Chair)
Pat Cirone, Region 10
Anne Fairbrother, Office of Research and Development
Harvey Simon, Region 2
Daniel Vallero, Office of Research and Development

Issue Paper Author

Bryan Norton, Georgia Institute of Technology

Issue Paper Peer Reviewers

Kent Thornton, FTN Associates (Lead Reviewer)
Robert Bachman, Maryland Department of Natural Resources
Thomas O'Connor, National Oceanic and Atmospheric Administration

FOREWORD

Publication of these issue papers is part of a long-term effort to develop Agencywide ecological risk assessment guidelines for EPA. Preliminary work on guidelines development began in 1989 and included a series of colloquia sponsored by EPA's Risk Assessment Forum to identify and discuss significant issues in ecological risk assessment (U.S. EPA, 1991). Based on this early work and on consultations with EPA's Science Advisory Board (SAB), EPA decided to develop ecological risk guidance in step-wise fashion, starting with definitions of terms and concepts and continuing with the development of source materials for the guidelines. The first product of this effort was the report *Framework for Ecological Risk Assessment* (U.S. EPA, 1992a,b), which proposes basic principles and terminology for the ecological risk assessment process. Since that time, other materials have been developed, including suggestions for guidelines structure (U.S. EPA, 1992c), ecological assessment case studies (U.S. EPA, 1993; U.S. EPA, 1994), and the issue papers contained in this report.

The issue papers were developed to help provide scientific and technical information that EPA scientists could use along with other materials to develop ecological risk assessment guidance. EPA did *not* ask the issue paper authors to provide guidance, to write a "cook book" or "how to" of methods, or to resolve differences between the papers in terminology or approach. Rather, EPA asked for a document of limited scope that would highlight important principles and approaches relevant to the ecological risk assessment framework that EPA scientists should consider in preparing guidelines. Synthesis and integration of the issue papers, framework principles, case studies, and other materials are deliberately reserved for the guideline writers and for subsequent peer reviewers and public commenters. The process of guideline development emphasizes peer review and consensus-building, as evidenced by the many experts from academia, industry, consulting firms, and State and other Federal agencies who have participated in the review of the issue papers and other guidelines source materials.

Compilation of ecological risk assessment guidelines for EPA is a challenging and complex task. We do not expect the first guidance document to answer all the questions or problems confronting the ecological risk assessor. However, we do think that the guidelines will be an important next step in an evolving process of improving the quality and consistency of EPA's ecological risk assessments. Such guidelines should also be useful to other Federal and State agencies, academic institutions, industry, environmental groups, and professional organizations.

William P. Wood, Ph.D.
Executive Director
Risk Assessment Forum

PART I. INTRODUCTION TO THE ISSUE PAPERS

1. BACKGROUND

In 1990, EPA embarked on a long-term effort to develop Agencywide ecological risk assessment guidelines. Drawing on experience with EPA's human health guidelines and advice from experts in ecological risk assessment and related disciplines (ecology, ecotoxicology, chemistry, etc.), as well as other sources (National Academy of Sciences, EPA's Science Advisory Board), EPA has developed a range of materials that can be used by EPA scientists in developing guidance. Specific documents include:

- Summary Report on Issues in Ecological Risk Assessment (U.S. EPA, 1991)
- Peer Review Workshop Report on a Framework for Ecological Risk Assessment (U.S. EPA, 1992a)
- Framework for Ecological Risk Assessment (Framework Report; U.S. EPA, 1992b)
- Report on the Ecological Risk Assessment Guidelines Strategic Planning Workshop (U.S. EPA, 1992c)
- A Review of Ecological Assessment Case Studies From a Risk Assessment Perspective (U.S. EPA, 1993)
- A Review of Ecological Assessment Case Studies From a Risk Assessment Perspective, Volume 2 (U.S. EPA, 1994)

The latest guideline source materials to be developed are the issue papers contained in this report, which help provide a bridge between the minimal descriptions of the ecological risk assessment process provided by the Framework Report and the future ecological risk assessment guidelines. As discussed in this chapter, EPA selected the issue paper topics around areas relevant to the ecological risk assessment process and asked the authors to provide a limited, focused review of principles and approaches in these areas. Further integration of the issue papers and resolution of EPA science policy issues are deliberately left to the EPA scientists who will prepare the Agencywide ecological risk assessment guidelines.

This chapter introduces the issue papers in the context of the other major source materials for guideline development, including the Framework Report and the case studies. The nature of the future ecological risk assessment guidelines are also discussed.

2. EPA'S FRAMEWORK FOR ECOLOGICAL RISK ASSESSMENT

EPA's ecological risk assessment framework (U.S. EPA, 1992b) describes basic elements, or a framework, for evaluating scientific information on the adverse effects of physical and chemical stressors on the environment. The Framework Report offers starting principles and a simple, flexible structure as guidance for current ecological risk assessments and as a foundation for future EPA proposals for risk assessment guidelines. The Framework Report was the product of an intensive 3-year effort beginning in 1989 that involved EPA scientists and experts from many different organizations. This section provides a brief overview of the activities that led to the Framework Report.

In 1989, EPA work groups began to develop endpoint-specific ecological risk assessment guidelines for aquatic populations, aquatic communities, and terrestrial populations. However, the SAB, in an initial consultation, recommended that rather than trying to develop comprehensive guidelines right away, EPA should start by defining the basic principles and process for ecological risk assessment. This provided the initial impetus for developing the Framework Report.

In 1990, EPA work groups and the National Academy of Sciences' (NAS) Committee on Risk Assessment Methodology began to study the 1983 NAS risk assessment paradigm (NRC, 1983), which provides the organizing principles for EPA's health risk guidelines, as a possible foundation for ecological risk assessment. Late in 1990, early drafts of the Framework Report received preliminary peer comment. In February 1991, while the draft Framework Report was in the preliminary stages of peer review, the NAS sponsored a workshop in Warrenton, Virginia, to discuss whether any single paradigm could accommodate all the diverse kinds of ecological risk assessments. There was a consensus that a single paradigm is feasible but that the NAS health paradigm would require some modification to fulfill this role (NAS, 1993).

In April 1991, EPA sponsored a strategic planning workshop in Miami, Florida. The structure and elements of ecological risk assessment were further discussed. Finally, in May 1991, EPA invited experts in ecology and ecotoxicology, including some attendees at the February and April meetings, to a peer review workshop on the draft Framework Report. EPA then integrated information, concepts, and diagrams from the previous discussions and workshop reviews with EPA practices and needs to propose a working framework for interim use in EPA programs and for continued discussion as a basis for future risk assessment guidelines.

The basic structure for EPA's ecological risk framework is shown in figure 1-1. The framework recognizes three distinct phases in the risk assessment process: problem formulation, analysis, and

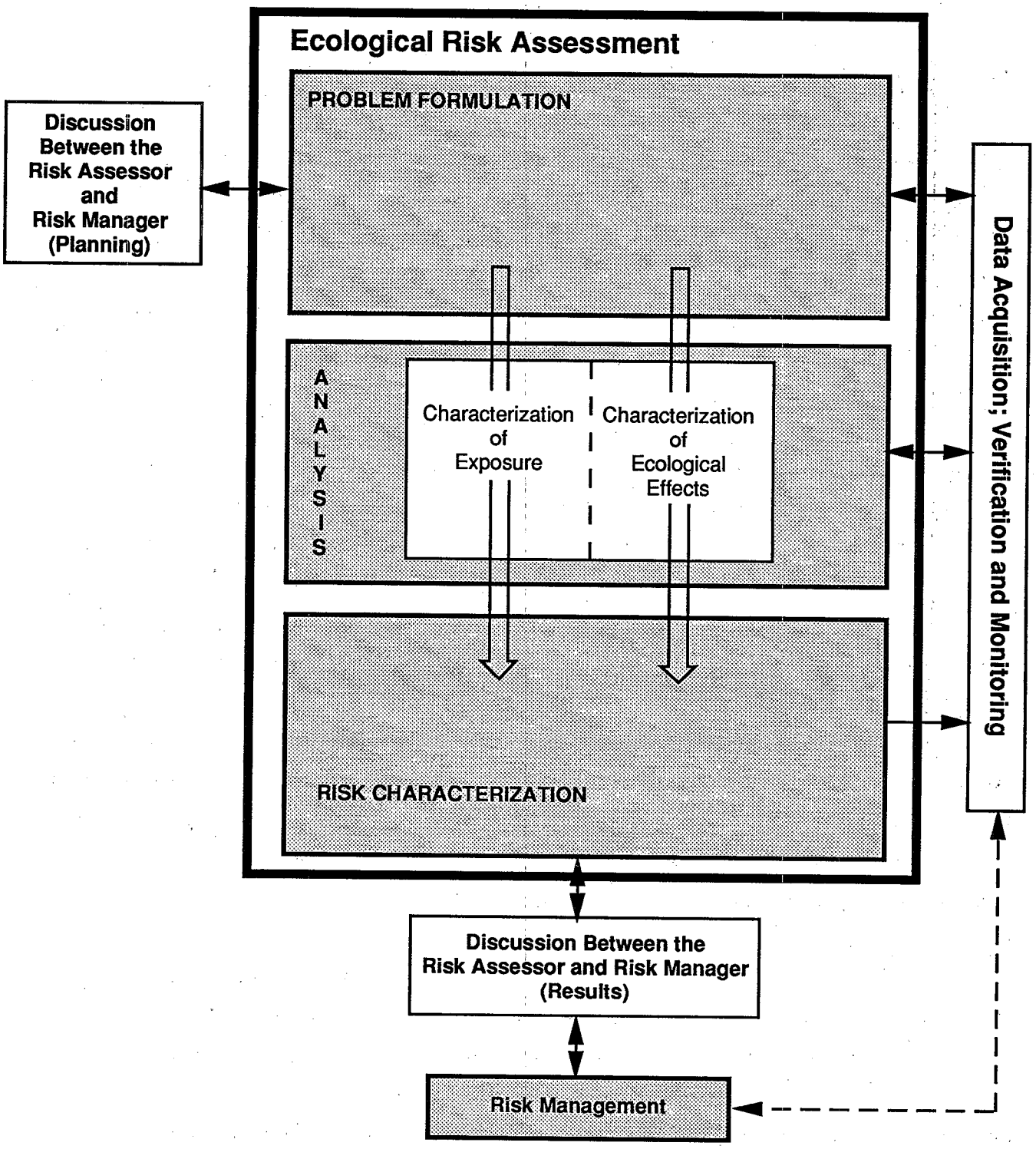

Figure 1-1. **The framework for ecological risk assessment (U.S. EPA, 1992b). The ecological risk assessment framework is the product of a series of workshops and reviews that involved both EPA and outside scientists. While the Framework Report has been a critical first step in developing ecological risk assessment concepts, evolution of the framework concepts is expected and encouraged.**

risk characterization. The problem formulation phase involves evaluating the potential stressors, ecological effects, and ecosystems at risk, selecting appropriate endpoints, and developing hypotheses that will be the focus of the assessment. Data on ecological effects, stressor-response relationships, and exposure to stressors are evaluated in the analysis phase. In risk characterization, exposure and effects information are integrated, principal uncertainties are summarized, and the ecological significance of observed or predicted changes is evaluated.

Note that the framework was developed with both chemical stressors and physical stressors (e.g., habitat alteration, hydrologic change, etc.) in mind. Biological stressors such as introduced species were not included because, while the general principles described in the framework may be helpful in addressing risks associated with these organisms, the capacity of such organisms for reproduction and biological interaction introduced additional considerations that could not be addressed in the Framework Report.

3. ECOLOGICAL ASSESSMENT CASE STUDIES

EPA's ecological risk framework is currently being used, evaluated, and tested both inside and outside of the Agency. One approach used to evaluate the framework was to develop sets of peer-reviewed case studies. These cases offer scenario-specific examples that can contribute to the development of future ecological risk assessment guidelines.

There are three sets of case studies. The first set, published in 1993 (U.S. EPA, 1993), contains 12 case studies that explore the relationship between the process of ecological risk assessment and common approaches used by EPA (and others) to evaluate adverse ecological effects. The case studies are wide-ranging in scope, representing a variety of ecosystems, ecological endpoints, chemical and nonchemical stressors, and programmatic requirements within the Agency. The case studies were evaluated at peer review workshops as to whether they (1) effectively addressed generally accepted components of an ecological risk assessment—problem formulation, analysis, and risk characterization—or (2) addressed some but not all of these components or, instead, (3) provided an alternative approach to assessing ecological effects.

A second set of five peer reviewed case studies (U.S. EPA, 1994) expands the range of the first case study set by including different kinds of stressors (radionuclides, genetically engineered organisms, and physical alteration of wetlands) and programmatic approaches (Pre-Manufacture Notice assessments under the Toxic Substances Control Act and the EPA's Environmental Monitoring and Assessment Program). The third set of case studies, now in an early stage of development, were jointly sponsored by the Risk Assessment Forum and EPA's Office of Water to explore the applicability of framework concepts to larger spatial scales. In particular, the Office of Water is interested in using a framework approach for conducting watershed assessments.

It is important to note that, although the case studies have been structured as described in the Framework Report, most were not originally planned and conducted as risk assessments. In addition, while these cases are representative of the state of the practice in ecological assessments, they are not necessarily models to be followed. Rather, they may be used to attain a better understanding of ecological risk assessment practices and principles. Three of the case studies are used as examples in the Conceptual Model Development issue paper, and others may be useful in the development of future EPA ecological risk assessment guidelines.

4. ECOLOGICAL RISK ASSESSMENT ISSUE PAPERS

Readers should understand that the issue papers do *not* constitute EPA guidance for conducting ecological risk assessments. Rather, they are important source documents for scientific and technical information that will be used by EPA scientists in preparing Agencywide guidelines. The issue papers were prepared to provide a bridge between the skeletal structure provided by the Framework Report and the more fully developed concepts needed for Agencywide risk assessment guidelines. EPA initially proposed the issue papers as part of the guidelines development process in a March 1992 meeting with the SAB. The SAB endorsed the use of issue papers and suggested several of the topics that appear in this volume. The issue papers in this report are the product of a 2-year collaborative effort involving EPA scientists and more than 40 independent qualified authors and peer reviewers.

Ideally, the intent of the issue papers is to cover basic principles spanning the full range of possibilities encountered in ecological risk assessments, including a variety of ecological systems (e.g., aquatic, wetland, terrestrial), stressors (chemical, physical, or biological), levels of biological organization (e.g., individual, population, community, ecosystem, or landscape), and spatial or temporal scales. Given this complexity and breadth, EPA elected to make the issue papers more manageable by carefully defining their scope to include guidance-relevant information. Thus, when evaluating the issue papers, readers should keep in mind the following guidance given the authors:

- The scope of the issue papers was limited to information needed for EPA's first guideline document (see section 5). As a result, they were not intended to address all areas or to be comprehensive literature reviews or monographs on selected topics. EPA suggested a 60-page limit for each document.
- The content of the paper was to focus on principles and approaches, not descriptions of methodologies.
- Authors were asked to identify existing, accepted ecological and risk assessment practices, as well as state-of-the-art approaches, significant information gaps, and future research needs.
- The diversity of viewpoints among the authors was considered advantageous to future guideline writers. Therefore, authors were not asked to reach consensus on terminology or technical issues. Synthesis of concepts and integration of issue paper principles were deliberately reserved for EPA guideline writers and for future peer panel discussions.
- To further assist guideline writers, the authors were asked to use the framework ecological risk assessment process as a starting point for organizing the issue papers, even though this

would result in a certain amount of redundancy or conflict among the papers. Moreover, the authors were free to suggest modifications to the framework as necessary.

- Authors were encouraged but not required to use examples or case studies in their issue paper for illustrative purposes.

Some of the issue papers cover specific portions of the ecological risk assessment process as defined by the framework, while others discuss areas that are relevant to the entire process. The focus of each of the nine papers and its fit to the framework process is briefly described below (see figure 1-2).

Ecological Significance

This paper approaches two basic aspects of ecological significance: societal evaluations of what is ecologically important, and scientific questions concerning what constitutes an ecologically significant change and how that relates to statistically defined significance. Themes include the importance of temporal, spatial, and organizational scale, selection of appropriate ecological (assessment) endpoints, and the concepts of relative risk and ecological sustainability. With respect to the framework process, the ecological significance paper covers topics relevant to all phases of the process, including problem formulation, analysis, and risk characterization. In addition to scientific considerations, this paper discusses societal aspects important to the risk manager.

Conceptual Model Development

Conceptual model development is the culmination of the problem formulation stage of an ecological risk assessment. However, this paper describes more than just the conceptual model. Rather, it covers the major steps in problem formulation leading to the conceptual model, including classifying the problem, summarizing available data on stressors and ecosystems at risk, and determining assessment and measurement endpoints. Principles are illustrated by case studies that are discussed throughout the paper.

Characterization of Exposure

While this paper is primarily concerned with how exposure is characterized in the analysis phase, it also discusses exposure considerations in problem formulation and linkage to risk characterization. It covers both chemical and physical stressors as well as disturbances and

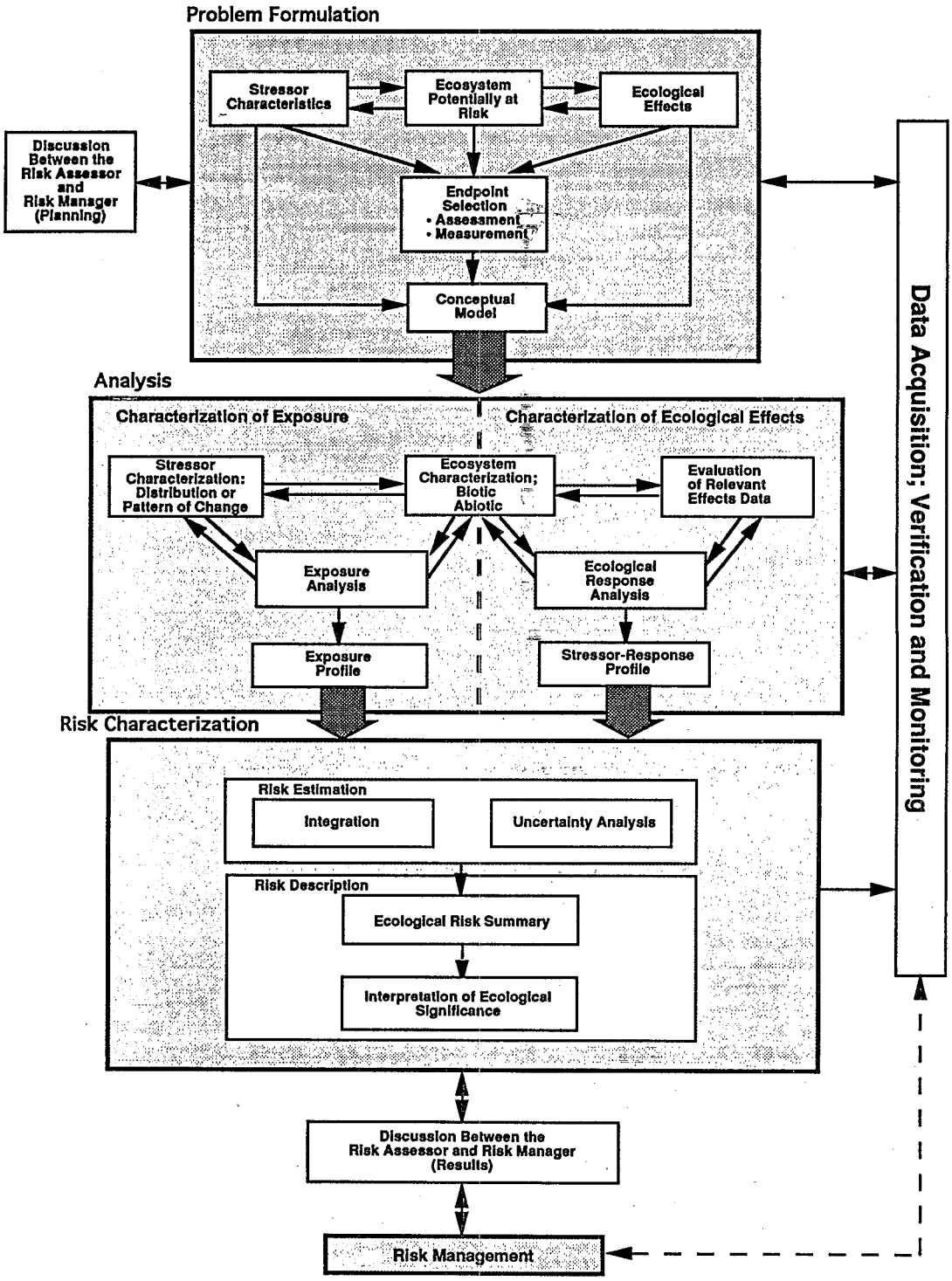

Figure 1-2. Expanded framework diagram (modified from U.S. EPA, 1992b). Some issue papers correspond to specific portions of the ecological risk assessment process, while others address "cross-cutting" issues.

deletions (harvesting, etc.) and suggests some exposure-relevant terms that differ from those proposed in the Framework Report.

Effects Characterization

This paper describes the effects portion of the analysis phase and complements the previous paper. Topics explored include stressor and assessment endpoint relationships, patterns of response to one or more stressors, tools for ecological response analysis relating assessment and measurement endpoints, and approaches to developing stressor-response profiles.

Biological Stressors

As noted above, the Framework Report did not discuss biological stressors (introduced species and genetically engineered organisms). While this paper does not propose an alternative framework for evaluating the risks posed by biological stressors, it outlines the key differences between biological and nonbiological stressors and summarizes information on the survival, proliferation, dispersal, effects, and uncertainties associated with biological stressors. Recovery from biological stressors is also discussed.

Ecological Recovery

There is very little discussion of the concept of recovery of ecological systems in the Framework Report. This paper describes considerations of recovery in problem formulation, basic factors influencing recovery (physical and biological as well as economic and social); ecological concepts related to recovery; and reference site selection, uncertainty and ecological significance. Recovery from anthropogenic and natural disturbances is discussed.

Uncertainty in Ecological Risk Assessment

Uncertainty is a fundamental concept that separates risk assessment from impact assessment. Evaluation of uncertainty occurs in all phases of the risk assessment process, and this paper describes uncertainty considerations important for problem formulation (e.g., model structure, scale, and ecosystem and stressor characterization), analysis (e.g., statistical analysis of data, extrapolation, and modeling), and risk characterization. Integration of uncertainty into the decision-making process is also described.

Risk Integration Methods

This paper describes a number of techniques for integrating exposure and effects information in the risk characterization phase of an ecological risk assessment. After briefly discussing risk characterization considerations in problem formulation and analysis, this paper covers a range of risk integration methods, including empirical models, process (mechanistic) models, and physical and experimental models. Other considerations such as uncertainties, natural variability, multiple stressors, and recovery are also included. Components of an effective risk summary are described. However, interpretation of ecological significance was covered by another issue paper, and a thorough discussion of risk communication was beyond the scope of this paper.

Ascertaining Public Values Affecting Ecological Risk Assessment

This paper was originally part of the ecological significance paper. It surveys available methodologies for identifying and measuring societal values that may be important for ecological risk decision making. Both market analysis techniques and other more nontraditional approaches to evaluating social values are discussed.

5. NEXT STEPS: ECOLOGICAL RISK ASSESSMENT GUIDELINES

Preparing Agencywide ecological risk assessment guidelines will be a challenging task. This section describes the Risk Assessment Forum's historical role in developing guidelines, describes the general nature of guidelines, and discusses future steps in developing ecological risk assessment guidelines.

For several years, the Risk Assessment Forum has been developing risk assessment guidelines for human health effects. In 1986, EPA issued guidelines on five topics, including cancer, developmental toxicity, and exposure assessment (51 *Federal Register* 33992-34054, 24 September 1986). Although EPA had issued guidance for cancer risk assessment 10 years earlier (41 *Federal Register* 21402, 1976), the 1986 guidelines substantially enlarged the scope of EPA's formal guidance by covering additional health topics and by covering all areas in much greater depth. Each of the guidelines was a product of several years of discussion and review involving scientists and policymakers from EPA, other Federal agencies, universities, industry, public interest groups, and the general public.

The guidelines developed by the Risk Assessment Forum are intended to improve the quality and consistency of Agency risk assessments. Guidelines can also establish Agency policy on difficult or controversial scientific issues. However, there are several ways in which the scope of the guidelines is limited.

- Since the guidelines are intended to have broad applicability across the Agency, application of the guideline principles for specific purposes is the responsibility of individual offices or programs. For example, the Superfund Public Health Evaluation Manual draws in part on the generic Risk Assessment Forum guidelines, but is a separate highly detailed guidance document specific to the Superfund program.
- Guidelines focus on defining scientific principles for analyzing the data and information available to risk assessors. They do not address research or test methods, nor do they cover the social, political, economic, or legal aspects that are part of the risk management process.
- The primary audience for the guidelines are Agency risk assessors. Since the guidelines assume that the audience is generally knowledgeable in the field, extensive background information on basic concepts is not provided.
- These guidelines are not regulations or legal requirements. However, others outside of EPA, such as Federal and State agencies, academic institutions, industry, environmental groups, or professional organizations, may use the guidelines and even adopt them.

- Guidelines rely heavily on the scientific judgement of the risk assessor. There are few rules that must be followed but many areas where considerations and criteria are provided for the risk assessor to consider.

Starting with these general principles, there are a range of options available for the proposed ecological risk assessment guidelines. The approach described here, based on past experience and discussions, will continue to evolve with time and experience. Because of the breadth and complexity of ecological risk assessment, the initial guidance document will not attempt to cover every type of ecosystem, stressor, level of biological organization, or spatial scale. Rather, the first guideline will be process-oriented, expanding on (and, if necessary, modifying) the principles found in the Framework Report. The ecological risk assessment guideline will integrate and synthesize information from the issue papers, case studies, and other materials. Additional guidance on specific subjects may be provided in additional guidance documents. This proposed structure is consistent with the recommendation of the 1991 ecological risk guidance strategic planning workshop (EPA, 1992c) for a "bookshelf" of guidance documents, including a primary "road map" guideline that would provide general information and direct the reader to other volumes in the bookshelf as required. A similar structure was discussed at the recent peer review workshop on the issue papers.

Whatever structure is eventually adopted for the ecological risk assessment guidelines, it is clear that the guidelines will not be completed in one step. Rather, the process of guideline development will be iterative, with initial guidance followed by subsequent revision and expansion as more experience is gained. Aspects of the original Framework Report may be superseded by the first guidelines, which will in turn be modified by later efforts. At each step in the process, peer review and public comment will continue to play a critical role.

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PART II. THE ISSUE PAPERS

Issue Paper
on

ECOLOGICAL SIGNIFICANCE

Mark Harwell
Rosenstiel School of Marine and Atmospheric Science
University of Miami
Miami, Florida

John Gentile
Risk Assessment Forum
Environmental Research Laboratory
U.S. Environmental Protection Agency
Narragansett, Rhode Island

Bryan Norton
School of Public Policy
Georgia Institute of Technology
Atlanta, Georgia

William Cooper
Institute for Environmental Toxicology
Michigan State University
East Lansing, Michigan

Prepared for:

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1. INTRODUCTION

The *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) provides a construct in which the risks to the environment from a human activity can be assessed. The framework is appropriate for assessing potential or anticipated environmental risks (e.g., for comparing the environmental risks from alternate management policies) as well as existing environmental problems (e.g., distinguishing alternate causes of observed conditions). The central objective of ecological risk assessment is to gain an improved understanding of the magnitude and likelihood of adverse environmental responses to human activities, explicitly taking into consideration uncertainties and recovery potential, so that informed decisions can be made based on the best available scientific data and knowledge. A critical element in the risk assessment process calls for distinguishing environmental responses that matter from those that do not; that is, determining the *ecological significance* of the risk.

There is no intrinsic ecological threshold for the establishment of ecological significance, although many ecological issues are germane. Further, what is significant and what is acceptable can only be determined through decision making that takes place in the context of human values. By addressing a variety of ecological and societal issues that relate to the determination of ecological significance, this chapter provides the societal context in which ecological risk assessments must be performed. Because there are no simple formulas to follow, guidelines for risk assessors must identify the types of issues to be addressed and provide a road map showing how to traverse these complex issues to reach a scientifically defensible and environmentally responsible decision.

The determination of ecological significance is complicated because it is a "meta-issue" in ecological risk assessment; that is, it is relevant to several aspects of the risk assessment process. In problem formulation, for example, issues concerning significance are relevant when societal and ecological values are being factored into the selection of assessment endpoints (e.g., in discussions between the risk assessor and risk manager). Similarly, in risk characterization, interpreting the significance of change in the assessment endpoint (e.g., probability of risk) is of critical importance to the scientist, manager, and the public.

The objectives of this chapter are to present a working definition of ecological significance; explore the process for incorporating social values into a determination of ecological significance; identify criteria for evaluating ecological significance; propose a decision framework for using the criteria; illustrate the decision framework with case-study examples; and identify research needs that are critical to the further development of this process.

2. DEFINING THE SIGNIFICANCE OF ECOLOGICAL CHANGE

The issue of ecological significance is fundamental to the entire ecological risk assessment process. When gauging significance, the risk assessor must identify risks that merit attention from the array of possible ecological changes that could be associated with some human activity. The basis for making such determinations is apparent, even if the specific elements of ecological significance are quite complicated:

- Virtually all components and processes in the environment exhibit natural variability continuously and simultaneously on many time scales.
- Virtually any human activity will result in a change to some component or process in the environment although only anthropogenic changes that can be distinguished from natural variability are appropriate for further consideration.
- Only a small subset of detectable anthropogenic changes actually matter to the structure, functioning, or overall health of a particular ecosystem.
- The criteria for defining such a subset (i.e., determining what matters) are in part ecological questions involving basic issues of stress ecology and in part societal questions involving human values and perceptions.

2.1. Societal Preferences

Defining what is ecologically significant partly involves the judgment of society at large expressed through risk management, legislation and regulation, or some other mechanism. Consider, for instance, that human-managed or -dominated ecosystems may range from near-natural or pristine conditions (essentially as they existed in prehuman times) to the conditions in traditional zoos, where small segments of habitat or populations are preserved for public viewing. Between these two extremes lies a continuum of possible ecological states that could exist under particular management regimes (figure 2-1); for example, modern zoological parks with a diversity comparable to the natural environment; a monoculture crop or tree plantation; a forest maintained for periodic clear-cutting; a national park of biome remnants, such as tall grass prairie; and a wilderness area with rigid and exclusive management policies. Societal assignment of a use for a given landscape is highly determinative for defining ecological significance, even though the societal decision may not be explicit. The decision may imply many preferences concerning, for example, uniqueness of the habitat or species, recreational or aesthetic value, economic utility, or cost of restoration and management. Other aspects may relate to the legal and institutional framework in which such societal

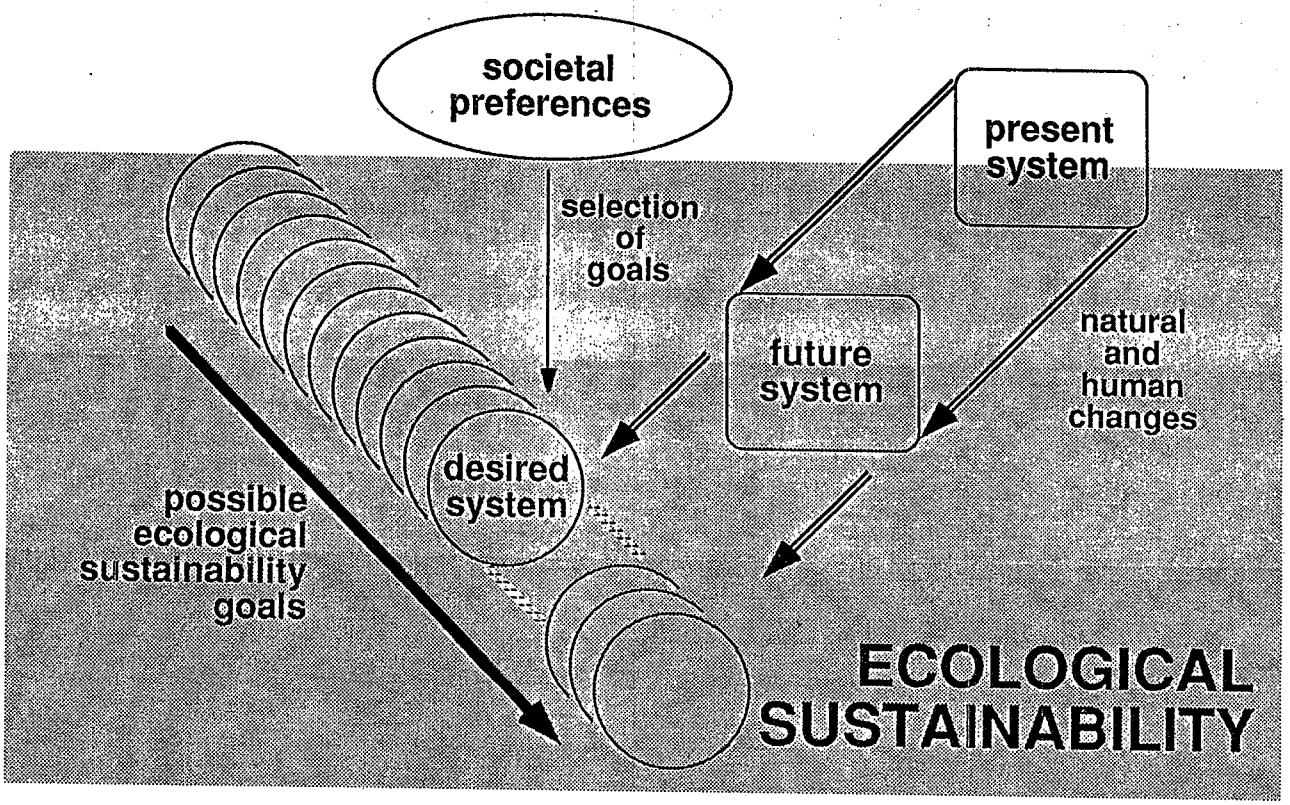

Figure 2-1. Ecological sustainability goals and societal decisions. There is a continuum of possible ecological states (ranging from pristine to highly modified) that could exist for a particular ecological system and that would be sustainable (i.e., maintained over a long time period). Societal decisions should identify where on this continuum a particular portion of the landscape or seascape should be. Appropriate environmental decision making should be designed to effect the change of an existing nonsustainable system to approach the desired sustainable condition. Societal preference shifts may discontinuously change the desired goals (from U.S. MAB, 1994a).

decisions are made and implemented, including the ownership and historical usage or preservation of the system, previous experiences with environmental catastrophes or successes, and even the personalities and priorities of individuals leading the dialogue from various perspectives. Overlaid on this is the potential for distinct shifts in societal preferences; for example, when a swamp that had been deliberately drained for agriculture or flood-control objectives is subsequently recognized as a valuable natural wetland.

2.2. Ecological Issues

The second major aspect of the significance issue is the ecological consideration, which requires that determinations about some issues are based on the nature of the ecosystem and its response and recovery characteristics with respect to particular stresses. Ecological issues include the nature of the stress and the ecosystem's experience with the stress. For instance, although fire itself is not a stress to a fire-adapted ecosystem, such as a grassland, the absence of fire or change in its frequency can become a stress. In contrast, fire could be catastrophic for an ecosystem that is not adapted to such a stress because the system only recently became exposed to it through human activities (e.g., tropical rain forests burned for land clearing and later abandoned).

It is important for the risk assessor to recognize that each ecosystem type has a different set of stresses to which it is vulnerable or to which it is indifferent. Similarly, each stress type may affect particular ecosystems differently. Thus the challenge in determining ecological significance is to identify those criteria or factors that are generic to all problem settings yet can be specified in sufficient detail to be useful in decision making.

2.3. Ecological Versus Statistical Significance

The third aspect that must be considered is the distinction between ecological and statistical significance. In scientific discourse, the term significance typically is assumed to mean the statistical confidence level at which a hypothesis is rejected. This statistical significance is based on the number of data points, the nature of their distribution, and the variability in the data. Statistical significance, however, is not necessarily the same as ecological significance, defined as changes in an ecosystem that are important in terms of its structure, functioning, or health. In contrast, a statistically significant change may or may not be ecologically important. For example, while tests of the species composition of the microbial communities of two ecosystems might show statistically significant differences, if redundancy in the functions performed by the species results in precisely the same

ecological condition regardless of composition, then the statistically significant differences are not ecologically significant.

Conversely, a determination that something is ecologically significant may not necessarily require that statistical significance be demonstrated. Indeed, reliance on stringent statistical tests would be contrary to the flexibility of the risk assessment paradigm. The ecological risk assessment process is multilayered, incorporating many different types of considerations into the decision-making process; these include nonecological factors, such as economics, and decisions such as whether to use a weight-of-evidence approach. Moreover, decisions often have to be made when the data are too few or too noisy for a specified statistical criterion of confidence to be met. For example, based on a wealth of information, it is clear that major changes in the global climate will result in ecological consequences that are highly significant; yet, given natural climatic and environmental variability and the intrinsic time lags in the global climate system, it would be unreasonable and irresponsible to wait for statistical proof at the 95-percent confidence level before making policy decisions.

2.4. Definition of Ecological Significance

As used in this chapter, ecological significance generally relates to the distinctions that must be made for determining (1) whether a change that is detected or projected in the ecological system or its individual components of concern is a change of importance to the structure, function, or health of the system and whether the change exceeds a variance estimate (i.e., the context of natural variability) and (2) whether such a change in the ecological system is of sufficient type, intensity, extent, or duration to be important to society. Both conditions are individually necessary and jointly sufficient for determining whether potential ecological change can be regarded as ecologically significant.

Several implicit assumptions are included in this definition:

- It is assumed that an assessment endpoint, which is a benchmark of the risk assessment, is the characteristic of the ecological system or its individual components of concern for which the significance of change is being evaluated.
- The assessment endpoints represent structures or functions that are important to the health and sustainability of the ecosystem.
- The assessment endpoints, at least in part, incorporate and reflect social values.
- Society is defined in its broadest terms to include the perspectives and values of scientists, resource managers, policy analysts, regulators, politicians, special interest groups, and the public at large.

- The explicit incorporation of variance into the definition reflects the need to introduce an envelope of thresholds above which a detected or projected change in the assessment endpoint can be evaluated. Defining an envelope of natural variability presents a distinct challenge, given the variability and stochasticity of many ecological systems.

3. SOCIETAL AND ECOLOGICAL VALUES

3.1. Interaction with the Risk Manager

The problem of ecological significance must be addressed at an early stage of the risk assessment/risk management process (i.e., beginning in the problem formulation stage). Unlike problems in human health risk assessment, assessment of ecological risk is not guided by widely accepted and easily identifiable societal values, such as maximizing individual welfare or reducing risk of disease and mortality. Identifying ecological values must be a part of the problem formulation component itself because observable changes in ecological systems often are not immediately classifiable as positive or negative. As a result, judgments must be made about whether change is significant (e.g., part of a trend rather than a stage of a cycle) and whether the change will affect some socially valued commodity or state. For example, when it was noticed in the 1960s that the waters of Chesapeake Bay were becoming more turbid, there was an immediate reaction from some individuals (e.g., recreational boaters) claiming an aesthetic loss and a few others claiming an economic loss (e.g., crab dippers); later it became apparent that the gradual changes were of much less consequence to recreational boaters. Only after years of discussion, scientific research, and experimental modeling was it determined that the trend toward turbidity was an ecologically significant change in the organization of the bay's ecosystem, probably caused by nutrient loading and the decline of filter-feeding oyster populations. It also took years for the change in water clarity to constitute a public problem of sufficient magnitude to justify a major policy initiative, titled "Save the Bay." As a result of this complex interaction of science, public debate, and value articulation, the Chesapeake Bay Regional Council established a goal for reducing nutrient loading into the bay's ecosystem.

General agreement that the environmental problem should be a high priority did not emerge separately from the determination of management goals or from the modeling of the system under management. In the conventional dynamic, science identifies the issues we face, but values determine whether we think the issues are a problem. To say that a certain change in an ecosystem has significance is to say that it warrants attention, and even this is a value judgment based on the common sense rule of "better safe than sorry." This approach recommends that data be gathered and interpreted on any important changes in the environment and that due caution be exercised in generating risks of irreversible change that may have strong negative impacts on our lives or the lives of future generations. Throughout the problem formulation process, someone must decide what changes are sufficiently important to monitor, when to recommend caution, and when to allow

changes to run their course. Ultimately, the process requires determining whether changes are consistent with the integrity of the system.

Ecological management is only vaguely analogous to health management because there is no clearly defined "patient" that experiences symptoms and seeks care. The description of some changes as insignificant and of trends across a landscape as healthy or maintaining integrity must be guided by particularly diffuse values that are experienced and expressed on distinctly different scales. Some of these changes are experienced as affecting individual welfare, but others are expressed on an intergenerational scale and have a communal aspect (e.g., protecting biological diversity for future generations). Assessing the scale at which to address a problem and whether trends are sufficiently expansive to threaten the health or integrity of an ecological system is a difficult undertaking. Yet for the concept of risk assessment to encompass ecosystems, significance must be considered an important criterion in determining which trends to monitor and where to encourage economic development.

3.2. Science and Public Interaction: A Process

Environmental decisions involve both societal values and science, much as do a physician's judgments regarding whether a particular patient is healthy or ill. This analogy between environmental management and health management can be instructive. For example, nobody would suggest that a definition of public health could be articulated without the general participation of scientists, physicians, and the public. Judgments about ecological significance and health call for the same degree of public and professional interaction. While the concepts of health and integrity are both normative and scientific, they are primarily public policy terms (Costanza et al., 1992). Likewise, by publicly debating the concept of ecosystem health, society defines what and how much environmental change is considered a problem. This conceptualizing process requires a blend of science, value articulation, and analysis. While concepts should be based on accepted scientific theories whenever possible, it must be remembered that such theories are anything but pure science.

Figure 2-2 illustrates the interaction of science, public discourse, and values identification in discussions of the scale at which to formulate a problem and measure success. Decisions about the scale at which to model and describe a problem, to define human impacts, and to formulate goals operate as a point of convergence for societal values, policy, and science. In the case of the Chesapeake Bay, formulation of the problem as one concerning water quality and identification of the stressor as nutrient loading from point and nonpoint sources were inseparable from a recognition that the ecosystem must be modeled at the scale of the entire watershed (Horton, 1987). Thus policy had

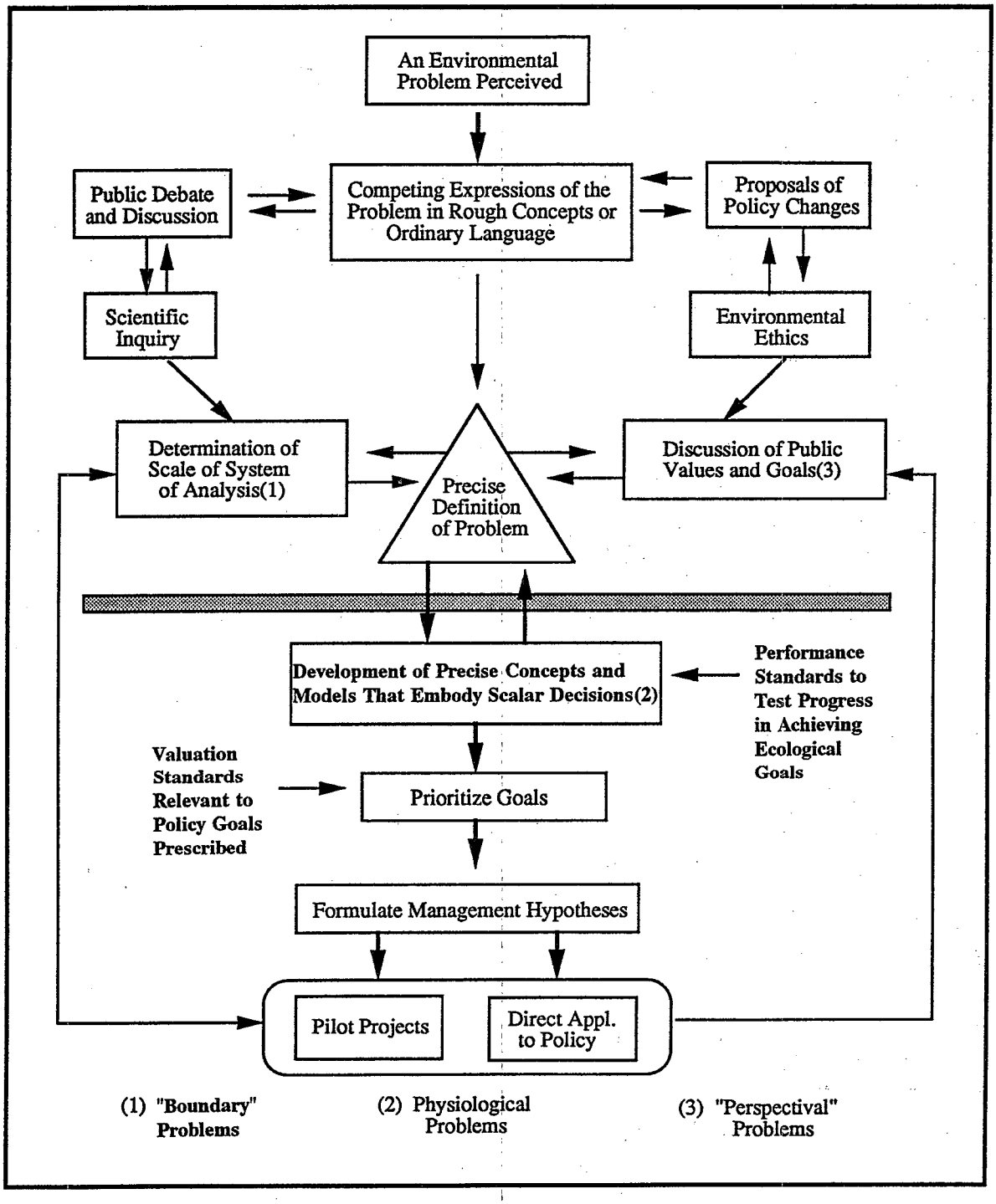

Figure 2-2. **The environmental policy process. Environmental problems are not clearly formulated when they first emerge in public discourse. Determining the proper scale at which a problem should be "modeled" requires an interactive, public process in which public values guide the scientific development of models. Once the problem is precisely defined and models developed, the process of experimentation with solutions can begin (Norton and Ulanowicz, 1992).**

to be established at the multistate or regional level. This case illustrates the importance of temporal and spatial scale as elements in decisions regarding ecological significance.

Because ecological systems can be modeled at many scales and on many levels, the interaction among scientists, the public, and managers must identify the scales at which ecological problems should be modeled. A prerequisite for decisive and effective policymaking is consensus among scientists and managers, with input from the public, regarding problem formulation and management goals. Indeed, policymakers must ask questions such as: Is ecological significance an issue in all risk assessments? What criteria can be used to make such a determination?

3.3. The Valuation of Natural Systems

All societies value the environment for the goods and services it provides. Most societies also find religious or spiritual value in the natural world; indeed, all of the world's major religious traditions, and many less prominent ones, explicitly recognize that the right to use natural resources carries with it an obligation to protect those resources for the future (Weiss, 1989). Such values can be difficult to separate into distinct categories. Thus, for example, when early environmentalists advocated setting aside land as national parks and other types of preserves, they did not distinguish between "use" and "nonuse" values and then assign them societal values, as do modern economists. Rather, they assumed that protecting certain lands would be the "right thing" to do and proceeded to act on available opportunities. Concerning natural systems then, it is best to think of societal values in terms of a shared continuum of emphasis—sometimes weighted toward use, sometimes toward longer-term spiritual and communal values—with different points on the continuum emphasized in different situations (Norton, 1991).

3.3.1. *Ecological Criteria and Values*

The integration of ecological criteria with economic criteria and societal values is fairly straightforward if one assumes that sustainability and survivorship are synonymous. For this to be true, however, one must assume a time domain of many generations (i.e., intergenerational equity) and a social structure that does not elevate the individual's rights and privileges above those of society. The relevant issues (e.g., intergenerational equity, stewardship vs. ownership, willingness to pay) are conceptually well understood, and the fundamental problems involve the temporal and spatial domains of the control systems and the institutional perception of the role of the human species in the ecosystem.

Ecological systems are shaped by evolution and managed by a process of natural selection. The constraints that impinge on individuals can be arranged in a dominance hierarchy, with the physical characteristics of the environment constituting the most restrictive constraints in the natural hierarchy, since modifications of local patterns of geochemical cycles or weather patterns are made with difficulty and are energy intensive. Within the particular habitat, the ecological constraints involve the direct and indirect interactions (e.g., predation, competition) of the coexisting biological components. Many animal species have developed social structures that constrain individual behavior through social dominance (e.g., caste-like systems). Also, a number of human societies still barter with the excess production of their ecological landscapes. In these societies, individual behavior (e.g., concerning rights, privileges, ownerships) and individual survivorship are constrained from the top down. Although not explicitly acknowledged by such societies, survivorship ultimately depends on sustainability. In contrast, industrialized societies that are energy and material intensive have redefined the control hierarchy socially so that it effectively operates in the opposite direction. Such concepts as ownership of property, declarations of rights, and the championing of democratic governments all contribute to the sanctity of the individual. The emphasis of this social philosophy has resulted in the prevalent notion of "being at war with nature," which has resulted in the permanent and widespread alteration of many aspects of the physical dimensions of our ecosystems in the name of fulfilling individual expectations.

The outcome of the individual-based control strategy is the evolution of societies and interest groups within societies that differ significantly in the emphasis they place on different points on a continuum of shared values. Avoiding conflict between economic growth/development and environmental protection requires understanding their inherently different temporal and spatial scales. Maintaining biodiversity requires protecting critical landscape-scale dynamics over long time periods. The challenge is to integrate multiple uses within the landscape by encouraging diverse uses at the smaller, individual-user spatial scales. The maintenance of landscapes that support critical ecosystem structures and functions will require the harmonization of short-term individual goals with the long-term, intergenerational needs of society.

3.4. Ecological Sustainability

Sustainability has become a popular term, in part because it can convey important principles for managing resources, but also because the term can mean different things to different people. The term sustainable development, which is even more commonly used, also can vary in its meaning,

from referring to the utilization of only renewable resources at rates compatible with long-term ecological health to exploiting resources extensively (i.e., just short of causing system collapse). Risk assessment is concerned with all types of ecosystems, from extensively human-altered systems to ecosystems selected for intensive care and stewardship. Consequently, the focus of this discussion is on ecological sustainability as it calls for maintaining the ecological system at a defined level of quality and health (Lubchenco et al., 1991; Edwards and Reiger, 1990). In general, human-affected ecosystems could be established at any of a variety of points along a broad continuum, from, for instance, an area totally covered with cultivated plants and caged animals on display, to an ecosystem that resembles its state in prehuman history in all respects. Although neither extreme is usually a realistic nor appropriate possibility, they indicate the scope of the continuum (see figure 2-1). Exactly where along the continuum a target is selected for ecological sustainability must be decided consciously, with attention to both ecological considerations and the realities of human systems.

The ecosystem's place on the continuum is particularly important for determining ecological significance. For example, stress-induced change in an already highly altered system, which in effect has been designated by society for human utilization and alteration, is quite different from a projected change in the health of an otherwise pristine, protected ecosystem, such as a national park or biosphere reserve. Similarly, maintaining a planting-harvesting rotation for a logging forest in which a limited number of species of trees are planted (e.g., ponderosa pine and Douglas fir plantings throughout the Pacific Northwest) would be considered ecologically not as significant as converting an old-growth, highly diverse forest into a deforestation-replanting cycle. Thus ecological significance assumes much greater importance for endangered, threatened, or otherwise especially valuable natural ecosystems for which ecological sustainability would be at risk from human activity.

Ecological risk assessment should not serve to preclude all future human activities or adverse impacts to the environment. Rather, when appropriately carried out through characterization of significance, risk assessment should be used to identify situations in which ecological sustainability is threatened. Defining ecological sustainability levels, however, must be consistent with ecological health principles. For instance, there must be a recognition that ecosystems constantly change, with natural variability occurring in relation to some average state as systems undergo succession or respond to long-term changes in the physical environment. Thus sustaining an ecosystem at some static condition is both ecologically inappropriate and impractical or impossible on a large scale. Similarly, just as ecological health must be evaluated in the context of particular ecological endpoints

that are defined in ecosystem-specific cases, ecological sustainability also must be defined in terms of specific ecological endpoints.

While these aspects of sustainability may seem simple enough, they often encompass conflicts related to biases that have not been resolved for most ecosystems. For instance, when management goals are defined ecologically (e.g., population levels of important species) rather than economically (e.g., yields of fish), they often are focused on maintaining particular population levels of selected species (e.g., a population of an aesthetically important wading bird). Separating natural fluctuations in species populations (e.g., caused by interannual variability in weather) from human-induced changes is not easily accomplished, however.

3.5. Significance, Sustainability, and Endpoint Selection

An almost unlimited number of ecosystem properties could be measured to characterize a particular system's health. Thus, to reduce all possible ecosystem responses to stress to a manageable level, we must choose a subset of potential ecological endpoints (Limberg et al., 1986; Harwell and Harwell, 1989; Kelly and Harwell, 1989; Harwell et al., 1986). One of the critical considerations for making such determinations concerning ecological significance is the correspondence between the ecological (or assessment) endpoints for the ecosystem and the targets of stress-induced change. If none of the stress-induced changes (direct or indirect) in an ecological system involve the selected ecological endpoints, then the change has no ecological significance. If some endpoints are affected, however, then other considerations must apply for determining ecological significance.

Therefore the selection of ecological endpoints (assessment and measurement endpoints) is one of the most important steps in the ecological risk assessment process (U.S. EPA, 1992). Ecological endpoints are defined as explicit expressions of valued environmental attributes that can be defined in operational terms and are amenable to measurement or prediction. Endpoints should be selected based on discussions between the risk assessor and risk manager during the initial phase of the risk assessment process. The selection of endpoints involves the juxtaposition of societal and ecological values and perceptions, a process that is complicated by the lack of a consensus concerning the definition of ecological health.

In regard to the issue of significance in an ecological risk assessment, endpoint selection establishes what is ecologically important in the problem setting and in terms of the societal value of the ecological resource. Once this is established, the risk assessment can be focused on determining

the probability and magnitude of change (risk) relative to that endpoint and whether that change is significant.

The diversity of ecosystems, each with its own unique set of spatial, temporal, ecological, and stressor characteristics, constrains the development of general selection endpoints. Nonetheless, some general concepts (e.g., sustainability) are applicable for developing selection criteria.

3.5.1. *Sustainability and Endpoint Significance*

There are two distinct but interdependent aspects of determining significance in ecological risk assessments. The first (i.e., problem formulation) focuses on defining the role and significance of the ecological endpoints within the broader context of the sustainability goals, landscape designation, and uses that are relevant to the problem setting. The second aspect (i.e., the ecological significance framework as discussed in section 5) focuses on quantifying the magnitude of change in the ecological assessment endpoint, comparing this change with natural variability, determining the spatial and temporal extent and patterns of change, and estimating the probability and time for recovery or reversibility.

Figure 2-3 schematically illustrates the relationships among ecological endpoints, sustainability goals, and landscape designation and usage. This concept proposes that the ecological endpoints used to assess risk must be placed within the context of the landscape's ecology in regard to sustaining the desired uses of the landscape over intergenerational time scales. This contributes in part to answering the "so what" question in risk assessment and is a first step toward addressing the question of ecological significance.

Although no single model exists for linking endpoints, landscape use, and sustainability, there are provisional criteria that can be used for this purpose. Relevant questions would include, for example: Does this problem setting pose a potential risk to (1) unique or critical components of the landscape (e.g., fragmentation, migratory corridors), (2) important populations (e.g., keystone, endangered species), (3) distinctive habitats (e.g., wetlands), or (4) the designated uses of the landscape? Do the endpoints chosen for the risk assessment adequately reflect these concerns? Is there the appropriate degree of connectiveness between the endpoints and the broader landscape ecology to address sustainability? Do the endpoints adequately represent and measure system redundancy, which is important in assessing ecological significance?

It is unlikely that all these issues will be important in every risk assessment. Nonetheless, the risk assessor and risk manager must be cognizant of the issues and must recognize when they are

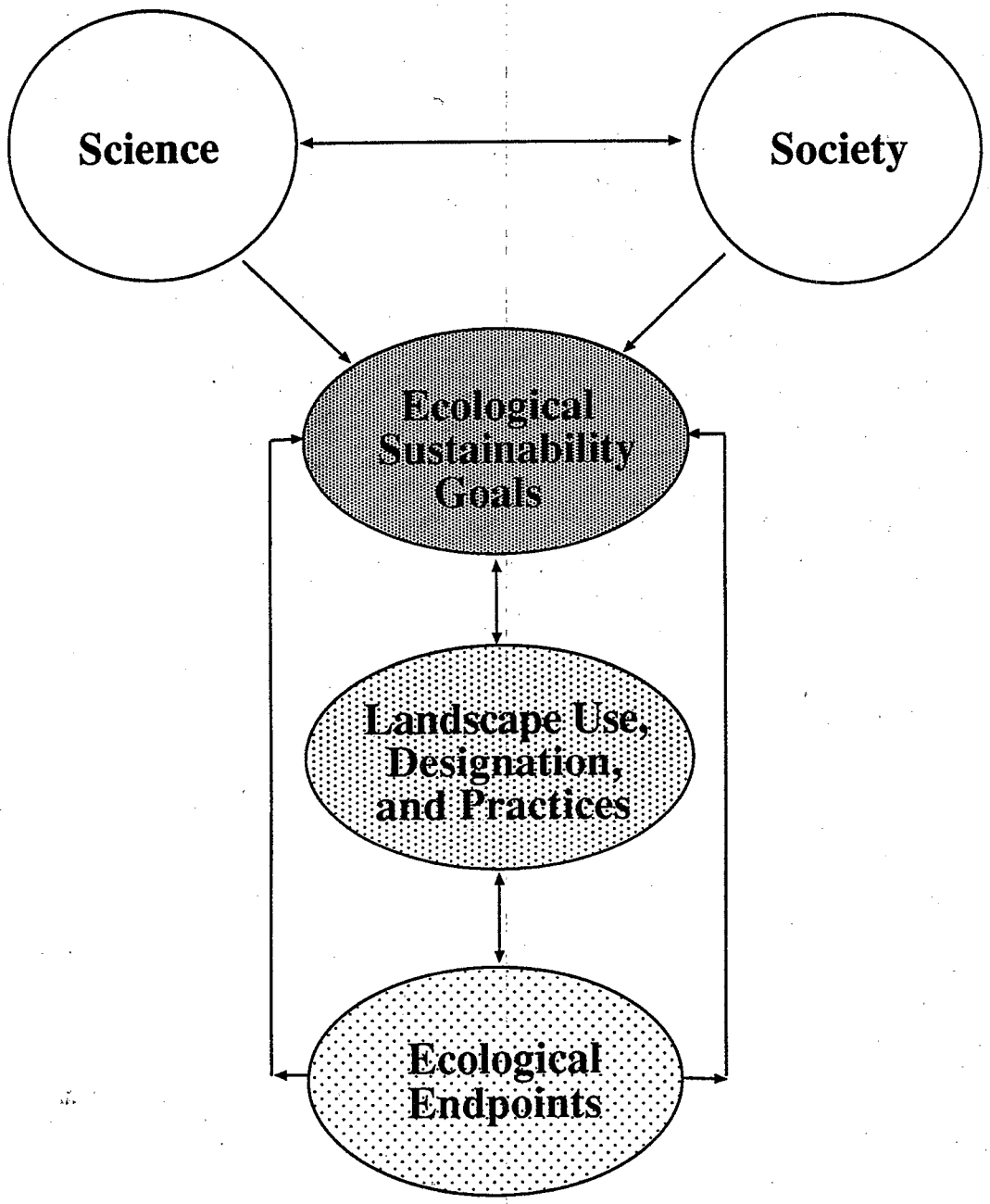

Figure 2-3. Interdependency of sustainability goals, landscape uses, and the endpoints used to establish benchmarks in ecological risk assessment.

relevant to a particular problem. Further, it is unlikely that quantitative linkages can be made between endpoints and sustainability for all situations. Instead, inferences based on accepted ecological theory and practice coupled with the weight-of-evidence and professional judgment often will have to suffice. Importantly, however, the prevailing view of ecological connectiveness must not be used as a rationale for assuming that all landscape properties are related and thus linked to sustainability (i.e., that all change is significant and unacceptable). All ecosystem structures and functions are not equally weighted; some are far more important to maintaining a sustainable ecological state (operationally defined) than others. Thus determination must be made on a case-by-case basis when developing the theoretical foundation for ecological significance. This point is developed further in the next section.

4. CRITERIA FOR DETERMINING ECOLOGICAL SIGNIFICANCE

4.1. The Nature of Ecological Change

Defining ecosystem health is by no means simple or straightforward. In contrast to organisms, ecological systems are less robustly defined, their dynamics are inherently less tractable, and their state is not easy to fully characterize. Further, ecosystems are constantly subject to stress, from both natural and anthropogenic sources.

Three concerns are fundamental: the nature of stress regimes, ecosystem responses to stress, and adaptation/recovery processes. The present understanding of stress, ecological response, and ecological recovery for most environmental stresses and for most ecosystems is incomplete and replete with uncertainties. In practice, a firm understanding is lacking of variability within and across ecosystems, which is significant because ecosystems are continuously in transition and continuously experiencing stochasticity. Moreover, it is difficult to establish appropriate measures of complex ecosystems for evaluating changes. These uncertainties notwithstanding, we do have a base of knowledge about ecological systems and how they are affected by human activities; thus we are not without the means to make appropriate environmental decisions. Lessons can be derived from natural stress-response relationships and from experiences with anthropogenic changes to the environment. The key is to identify factors that need to be systematically examined and criteria that need to be explicitly considered when determining the ecological significance of an anthropogenic change.

Norton (1991) proposed a risk topology that can use spatial scale, temporal scale, and reversibility as criteria for addressing ecological significance and relative ecological risks. These three factors (and, by implication, functional redundancy) are represented in figure 2-4, which plots the recovery time (as a measure of reversibility) of the impact of a decision against the spatial scale over which the change occurs. The criteria are consistent with the *Report of the Ecology and Welfare Subcommittee of the EPA Relative Risk Reduction Project* (U.S. EPA/SAB, 1991). Indeed, the decision square in figure 2-4 can be thought of as a formalization and simplification of an important recommendation of that subcommittee (U.S. EPA/SAB, 1991; Harwell et al., 1992). This "ecological risk decision square" can be useful for analyzing decisions that have long-term, difficult-to-reverse, and spatially pervasive impacts. Decisions that carry risks of outcomes that have all of these characteristics will cluster in the upper-left shaded region (i.e., the "critical region"). These decisions will require careful ecological study and thoughtful analysis of the social impacts of likely ecological changes and trends.

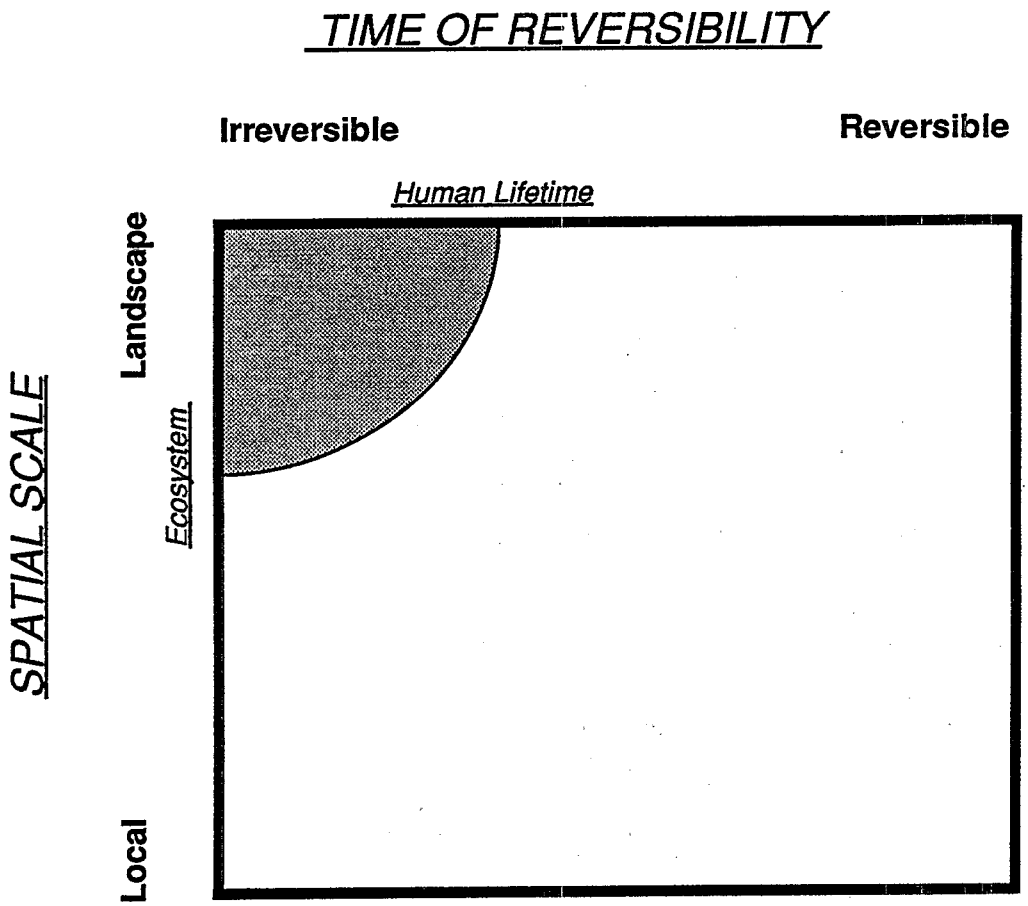

Figure 2-4. Risk typology: severity and reversibility (Norton, 1991).

The concepts and rationale embodied in figure 2-4 reflect the inherent complexity associated with the analysis of environmental problems. In this respect, the figure goes beyond the one-dimensional analysis provided by mainstream economics, which holds that all resources have adequate substitutes. This contention implies that there is a great deal of redundancy in the resource-producing system, thus placing minimal, if any, emphasis on irreversibility. This position of mainstream economists can be represented relative to the decision square by denying there are any decisions in the shaded or "critical region" and assuming that the risk decision space is continuous.

The proposed model introduces the concept of temporal and spatial scale into the decision process. In so doing, the model recognizes an initial need to establish the possibility that outcomes of a decision may present an irreversible threat to important ecological resources and to intergenerational values. This determination must be made before applying standard economic analyses that treat all costs and benefits (and increments and decrements of risks to humans) as expressible in monetary terms and as interchangeable, or fungible. This two-tiered approach to environmental decision analysis introduces complexity into the decision process. At the beginning, determining ecological significance based on the three criteria identifies decisions involving moral questions of intergenerational equity (because such decisions may greatly narrow options for future generations). The approach also introduces complexity at the decision stage, because one must apply criteria appropriate to the specific problem.

4.2. Criteria for Determining Ecological Significance

The definition of ecosystem significance sought in this chapter is based on a set of criteria, quantified where possible, but involving professional judgments on the part of scientists and managers. The definition would facilitate reaching informed decisions in the formative task of deciding what environmental changes are problems requiring management attention. While the goal of this chapter is to give more specificity to these criteria, we recognize that many details are problem and ecosystem specific. It would be naive to think that any specifications for the criteria could be stated with both precision and generality (Levins, 1966). Given that caveat, we propose the following criteria in determining the significance of predicted or measured change in the risk to one or more ecological endpoints.

4.2.1. *Ecological Scale, Complexity, and Endpoints*

Some ecological systems may be protected from significant stress-induced changes by internal processes of compensation or adaptation. For example, microorganisms may be capable of converting toxics into nontoxic substances by biochemical degradation. Also, toxic materials can be removed from sites of biological activity by the organisms themselves, through the uptake and sequestering of compounds in storage tissues. Yet ecological systems tend not to be adapted to rare, extreme events (e.g., freezing events affecting mangrove ecosystems) and may not be adapted to accommodate most anthropogenic stresses (e.g., inputs of toxic xenobiotic chemicals). Consequently, anthropogenic, chronic, or acute disturbances that do not mimic the frequency or nature of natural disturbances can alter the ability of a system to absorb damage and in turn alter basic properties of the ecological system. Qualitatively different responses by ecological systems to a disturbance will occur, depending on the frequency and novelty of the disturbance in the evolutionary history of the ecological system. Thus the same disturbance can have dramatically different consequences, depending on the particular ecological system—for example, the effect of fire on grassland ecosystems versus tropical rainforests. Similarly, a particular ecological system will likely respond differently to different disturbances. For example, the grassland may do well in the presence of fire but be devastated by overgrazing. The responses of ecological systems vary widely across systems and stresses. Insofar as an anthropogenic stress mimics a natural stress, stress-response relationships are likely to be established by the ecological system's adaptations to that type of stress. For novel stresses, however, such as those caused by human activities, stress-response relationships are unpredictable.

Depending on the specific problem, endpoints for ecological risk assessments can reflect the full range of ecological hierarchy and complexity. For example, species-level endpoints are characterized by population parameters, such as age distributions, genetic variation, and generation times; thus life-table parameters may represent the most relevant and integrative endpoints (Euler, 1970; Caswell, 1989). Biological communities have a development trajectory that is a function of their environment's characteristics and initial conditions. Additionally, communities are variable in structure over both temporal and spatial gradients. As a result, ecological endpoints should be structural characteristics (e.g., guilds) that reflect the stage of succession, the size of the system, and important processes (e.g., decomposition, nutrient cycles). Individual species as biological indicators will be relevant if they are tightly associated with a guild of particular concern. In terrestrial communities, the vegetation component significantly affects the faunal diversity, and the vegetation composition is generally determined by the soil type and the hydrologic conditions. Certain communities (e.g.,

grasslands) have additional determinants (e.g., fire and grazing). The temporal dimensions of community dynamics are usually measured in years and decades.

The ecosystem concept links the biotic community to the physical environment through the transport and fate of energy (trophic structure) and materials (biogeochemistry). For this level of organization, the shift in focus is from biotic structure (species associations) to ecosystem function. The degree of biotic influence on the fate and transport of materials depends on the stress. For example, phosphorus and calcium in the terrestrial system are largely controlled by physical processes, whereas nitrogen fate and transport are strongly modified by biologic processes.

The ecological endpoints at the ecosystem level of organization should be associated with critical functions rather than structural characteristics, unless they are uniquely associated with each other (i.e., no redundancy). The primary producer and the detritivore components are the most important aspects and should be given top priority.

4.2.2. *Natural Variability*

Ecological systems vary over time and space naturally, with or without human interference. That variability is differential over different components of the ecological system, different spatial scales, and different time periods. For example, a major change from one ecosystem type to another at a small scale may be highly significant ecologically. Yet if such patch-level changes occur throughout the landscape and over time, a mosaic at the landscape-level may develop that is in itself constant over time and without significant ecological changes. As another example, natural fluctuations in marine fish populations are often large, with intra- and inter-annual variability covering orders of magnitudes in population levels. This is natural, however, and a change in the population caused by some stress may merely be noise within the natural variability. By contrast, some populations are notably long-lived and constant in size over time, and a change in comparison with the natural low variability would be of particular ecological significance. For example, a transient 50-percent reduction in striped bass may not be ecologically significant, but a 50-percent reduction in the population of redwoods would be.

On what scale then should ecological system response to stress be measured? Clearly, no single scale can be selected exclusively. For example, photosynthesis functions at the cellular level on a time scale of minutes to hours, whereas the life cycle for the leaves is measured in months or even years, and that of the tree itself is measured in decades to centuries. Similarly, the population dynamics of the soil bacteria are extremely rapid and fluctuate wildly in time and space. By contrast,

the population dynamics of the forest's bears are long term and cover a large spatial extent. Responses of these populations must be measured on distinctly different time scales. If they are not, critical characteristics of population dynamics occurring out of synchrony with measurement intervals may be missed and the response of the population misinterpreted. Changes within the forest ecosystem can occur gradually over long time periods and may be affected by slowly changing external factors such as climate. These factors can operate over centuries to millennia.

Interactions between the spatial and temporal components of variability also need to be considered in assessing significance. For example, an extreme weather event may cause damage from freezing to some particular location but not to a nearby area because of microclimate differences (e.g., proximity to water, elevation, local wind conditions). Yet a synoptic event may lead to major impacts affecting all local areas, increasing the ecological significance and reducing recovery prospects. This is an important distinction in global climate change analyses: Any given weather station might experience an extreme cold or warm event, but if essentially all stations experience prolonged or frequent extreme events, the consequences are dramatically increased.

4.2.3. *Magnitude of Change*

The magnitude of stress-induced ecological change involves its intensity. The degree to which ecological endpoints change in response to a given stress is an inverse measure of the resistance of the ecosystem to that type of disturbance. Thus a highly resistant ecosystem would change only slightly in response to the same stress that would cause major displacement in a low-resistance ecosystem. Note that all evaluations of an ecosystem's resistance (or other measures of stress-induced response) must be made with respect to a particular stress or combination of stresses, because different stresses may elicit different responses. For this reason, one cannot accurately characterize a type of ecosystem as being intrinsically resistant, and resistance may be seen in one indicator of the ecosystem but not in another. How an ecosystem responds to perturbation and how readily it recovers (i.e., its resilience) concerns the stability of an ecosystem; like the ecosystem itself, stability can only be defined operationally.

4.2.4. *Spatial Extent of Change*

A factor relating to the intensity of a response is the spatial extent affected by the stress. Quite simply, ecological significance increases with the area affected. This is true for several reasons:

- A larger affected area includes a greater fraction of the total area of that ecosystem type, depending on the relative size of the affected area versus the size of the habitat type. For example, an activity that would adversely affect the last remaining undeveloped area of the barrier islands off the Atlantic Coast is more ecologically significant than an impact covering the same area but of a much more extensive ecosystem, such as the deciduous forests in the eastern United States.
- A larger affected area is likely to be subject to a greater number of other stresses, increasing the complications from stress interactions.
- A larger affected area is more likely to contain specific components of concern, such as a habitat for endangered species.
- The larger the impacted area, the more difficult the recovery.
- A larger affected area may involve landscape-level changes because many ecosystems may be altered by the stress.

4.2.5. *Recovery/Reversibility*

Recovery is the rate and extent that an ecosystem changes in response to the removal of a stress. Again, there are two components: one related to how *rapidly* the ecosystem recovers, the other to how *effectively* the ecosystem recovers. The temporal aspect is characterized as the ecological system's resilience, which is defined as the inverse of the length of time required for an ecosystem to return to its near-normal state. One cannot reasonably define this as a complete return to a preperturbed state because natural heterogeneity might preclude ever attaining that precise state. Moreover, it is possible that prior to the disturbance the ecosystem was not at steady state, even in the absence of human interferences, since properties of an ecological system may change over time. For example, diversity of a forest ecosystem will increase during the early stages of ecosystem development, then will decline in the middle stages of succession and increase again during the later stages. Thus comparisons should be made not to a single set of steady-state values for the ecological system, but to a mutable set of characteristics describing the trajectory of the undisturbed ecological system.

Along with resilience issues, recovery concerns questions about whether the ecosystem will ever return effectively to its predisturbance state or trajectory. It is possible that a complex ecological system, when subjected to particular disturbances, will become irreversibly transformed into another system with different components, steady states, and dynamics. This is a well-known characteristic of

many ecological systems. For example, deforestation in the coastal hills of Venezuela has changed soil structure, seed sources, and the local physical environment such that forests cannot grow again, even after the areas are abandoned by humans. This phenomenon also was seen in the irreversible loss of the great forests in Britain during the Neolithic period, when humans cleared land for agricultural production and energy resources. Perhaps these examples merely reflect an exceedingly long period of recovery for an ecological system; yet for the practical purposes of human interest, these examples of system change are permanent. Recovery of ecosystems is in part dependent on the characteristics of the stress (i.e., the disturbance regime, including the nature of the stress, its frequency, duration, and intensity) and in part on the history of the ecological system (e.g., the level of preadaptation to disturbance, past history of disturbance, and susceptibility of organisms within the ecosystem). An ecological system that has been subjected to repeated disturbances may tend to deteriorate over time because of loss of nutrient reserves or substrate. In other cases, recovery from repeated stress may be rapid if the important species complete their life cycles during the interim between disturbance events.

Characterizing the recovery of an ecological system presents the same challenges as characterizing an ecological system's response to stress—namely, identifying which endpoints to examine. For example, Is an ecosystem recovered when its pools of nutrients are back to the prestressed state? When a specific species has reestablished its population at a particular density? Or when the residues of a toxic chemical in sediments or in biological tissues have decreased to below some threshold? Just as an ecological system functions and responds to stress at widely differing rates, hierarchical levels, and spatial extent, it also recovers differentially. Selecting the appropriate suite of endpoints is not a trivial task, and there are substantial difficulties added in establishing an appropriate baseline for comparing the stressed ecological system. Also, natural heterogeneity and fluctuations again raise the issue of detecting signals from amid the noise of natural variations.

4.2.5.1. Redundancy

Ecological redundancy means that one ecological component can perform a similar function to another component. Redundancy can allow the ecological system to adapt or compensate for a stress-induced change such that no functional changes occur in the system. As a result, determining ecological significance relates only to the importance of the structural changes, without concerning any functional considerations. Thus, if a redundant species has no other particular value (e.g., one of many fungi involved in decomposition), then its loss would not be considered ecologically significant.

Conversely, loss of a species for which there is no functionally redundant counterpart would constitute ecologically significant change, irrespective of the direct value of the species to humans. In general, changes in an ecological system's biotic populations may or may not be of direct ecological importance, because redundancy or other compensatory mechanisms may mitigate adverse effects on ecological system processes; however, because changes in ecological system processes invariably result in changes in biological constituents, ecological significance is more likely to result.

4.2.5.2. Time for Change

A potentially important factor in ecological significance is the rate of change in the ecological system induced by a stress. If the stress response is sufficiently slow, the ecological effect may be compensated by changes in other ecological system components that ameliorate the consequences of the stress. Significantly, an ecological system typically consists of some processes occurring over short time periods simultaneously with other, often controlling processes that occur at rates that are orders of magnitude slower. This wide divergence in time constants means that ecological significance for a particular ecological component is in part a function of the rate of the change and the intrinsic time for the component. For example, a rapid change in a slow-time component is likely to have greater impact than a change occurring at a rate consistent with the natural time dynamics of the affected component. Conversely, an ecosystem is more likely to adapt to a slow change in a rapid-time component, with concomitant reduction in ecological significance.

Further, ecological systems have intrinsic time lags such that adverse responses from a stress may be delayed. This is important when distinguishing the long-term impacts of a stress from the immediately visible effects. Thus caution must be used in ecological risk assessments to ensure that important but time-lagged adverse effects are not discounted or not identified.

5. THE ECOLOGICAL SIGNIFICANCE FRAMEWORK

5.1. Risk Characterization

The ecological risk assessment process is designed to determine the likelihood of a change in ecological state or condition occurring as the result of incremental changes in the stress regime. Although the statement of risk measures the probability of change, it does not necessarily assume that a discussion about the significance of that change has taken place (i.e., by explicitly addressing the "so what" question). A discussion of ecological significance should be included in the description of risk communicated by the risk assessor to the risk manager and the public.

Currently, issues of ecological significance are addressed on a case-by-case basis using a weight-of-evidence approach that is founded on best professional judgment. The strengths of this approach recognize the unique, site-specific nature of each problem; the limitations of available data; and the value of the expertise and judgment of those most familiar with the problem. The limitations of the current approach include the lack of consistency in decisions; the absence of criteria for evaluating significance; the lack of a strategy and process for applying the criteria in a systematic manner; and the absence of an overarching context (e.g., sustainability) for interpreting the results.

5.2. Purpose and Assumptions

The purpose of the ecological significance framework (ESF) is to organize, logically and systematically, the criteria for determining ecological significance (figures 2-5a and 2-5b). The ESF concept does not assume that a simple algorithm exists that can be used to determine ecological significance for all problems. Indeed, each problem setting represents a unique combination of stressors, ecological receptors, and scales. Nonetheless, a process can be outlined that describes how the criteria can be organized to facilitate decision making concerning ecological significance. (Note that the illustration of tier 2 of the ESF avoids use of a flow diagram, because the emphasis on this level of the framework is flexibility.)

The purpose of the ESF is to assist the risk assessor in determining whether the measured or predicted change in one or more assessment endpoints is ecologically significant. The application of the ESF at this phase of the risk assessment process assumes that a suite of assumptions has been addressed and satisfied earlier in the problem formulation phase of the risk assessment.

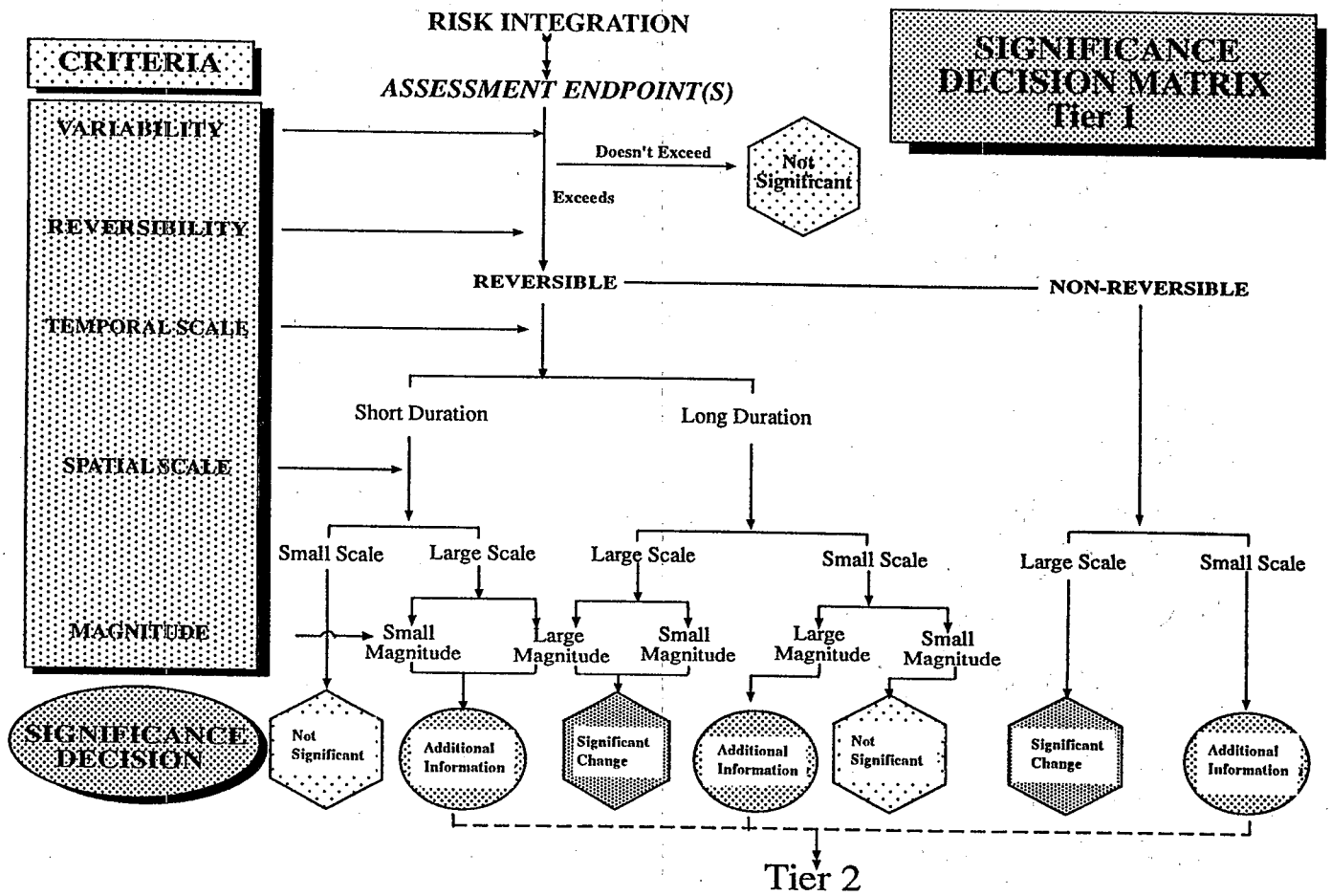

Figure 2-5a. **Tier 1 of the proposed ecological significance framework: a road map for using criteria to facilitate decision making concerning the significance of observed or predicted changes in ecological endpoints.**

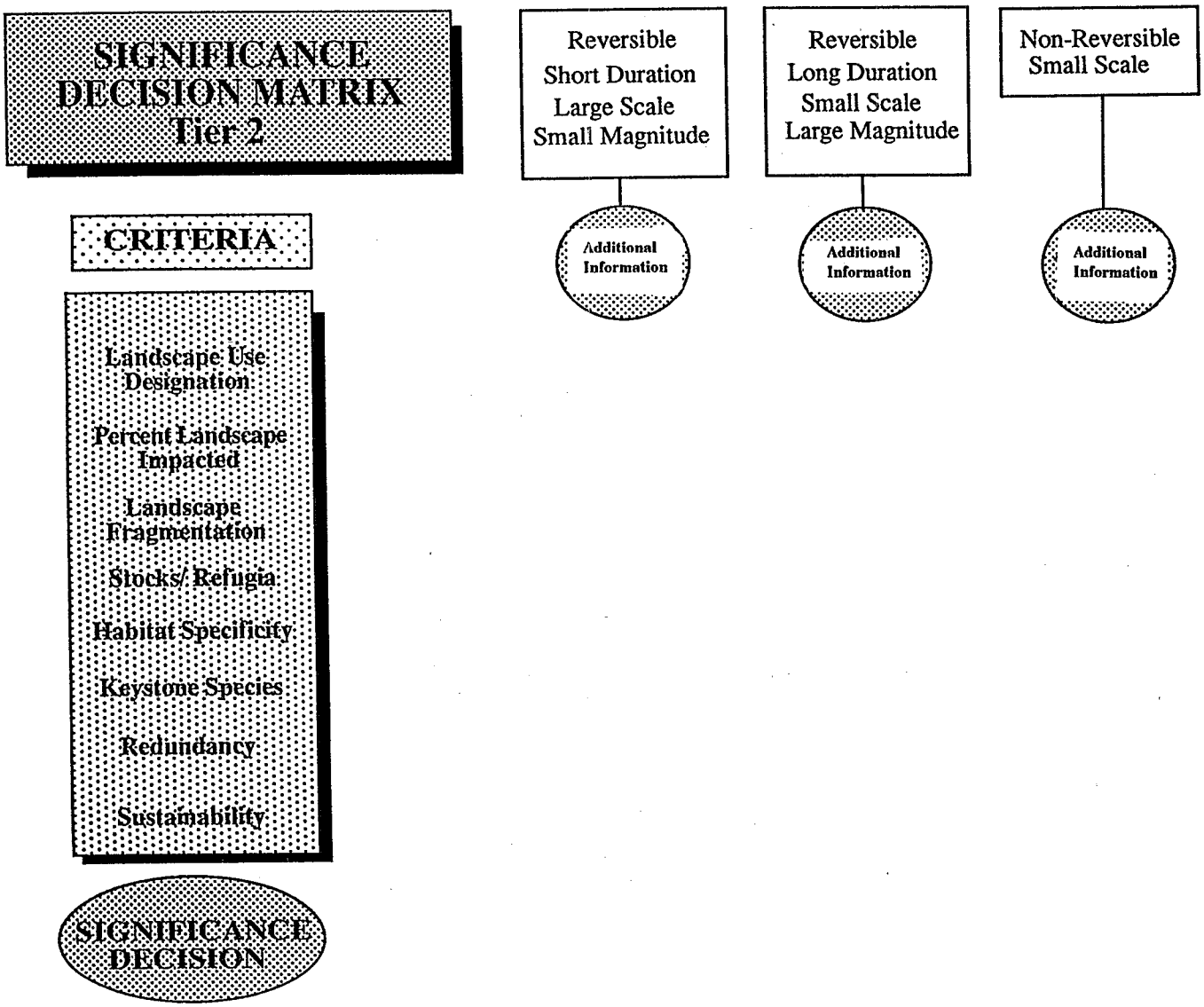

Figure 2-5b. **Tier 2 of the proposed ecological significance framework: a road map for using criteria to facilitate decision making concerning the significance of observed or predicted changes in ecological endpoints.**

5.2.1. *Problem Formulation Assumptions*

The two primary components of problem formulation are the selection of endpoints and the development of a conceptual model that describes the potential risks to the problem-specific ecological endpoints. The ecological assessment endpoints, which serve as the foundation of the ecological risk assessment, must incorporate ecological importance and social values, be defined operationally, and be measurable directly or through prediction (i.e., using models or statistics). Further, the ecological assessment endpoints should be closely linked to the long-term sustainable landscape use goals.

The conceptual model describes the spatial and temporal boundaries of the assessment; the exposure pathways and patterns for one or more stressors; the species, populations, communities, and ecosystems that are potentially at risk; and a relative ranking of potential risks for each ecological endpoint.

5.2.2. *Risk Integration Assumptions*

The risk integration phase of the ecological risk assessment process quantitatively combines exposure and effects profiles into a probabilistic statement of risk for one or more ecological assessment endpoints (see chapter 9 on risk characterization). The statement of risk describes the magnitude and direction of change in the assessment endpoint; the level of probability and uncertainty (preferably quantitative); and the appropriate stress-response models (e.g., ecotoxicological, population, systems based) for relating changes in the assessment endpoints resulting from incremental changes in exposure to one or more stressors.

5.3. The Significance "Road Map"

Determining the significance of the ecological changes requires careful and systematic application of the significance criteria to the changes in the endpoints used in the risk assessment. The proposed ESF functions much like a road map that suggests a direction from point A to point B without necessarily selecting the exact route. For this function, the ESF organizes the five primary significance criteria (i.e., tier 1 of the ESF: variability, reversibility, temporal scale, spatial scale, and magnitude/severity of the ecological effect) into a binary decision framework to facilitate a determination of ecological significance (see figures 2-5a and 2-5b). For some cases, additional information will be required.

Tier 2 of the ESF employs an additional suite of criteria. A specific set of generic routes for decision making is not delineated, however, because this would require the development of decision rules concerning the sequence and relative importance of each criterion, which is beyond the scope of this discussion.

The tiered ESF assumes that binary decisions can be made for each of the criteria. A more complex version of the ESF could be developed, presenting each of the criteria as scalar determinations. In the scalar version, each criterion would be assigned a range of zero to one, affording an opportunity for the risk assessor to rank the relative significance of risks from scenario-consequence analyses or remedial options. Again, it is important to point out that the proposed ESF is only one type of "road map" for use in decision making concerning ecological significance.

5.4. Significance Criteria

5.4.1. *Variability*

One of the more vexing issues in determining ecological significance is variability. The seminal question is simply, Are the observed changes in indicators and ecological endpoints distinguishable from the natural variability of the response being measured? That is, how much change in the endpoint is "natural" and to be expected given both temporal (e.g., intra- and inter-annual time scales) and spatial variability? Can these boundaries be delineated, and how do we know when the boundaries have been exceeded?

These questions are not trivial and require careful examination. Possible approaches include analysis of time series data; historical baselines; comparison of analyses of similar measurements at reference sites; and comparisons of laboratory control and reference data. Irrespective of the approach, data availability is always an important limitation. Time series data, while available for fisheries and some wildlife populations, are generally very limited. Confounding the issue of data availability is the highly variable nature of ecological data. Since ecosystems and their ecological components are not homogeneous in either space or time, one has to develop rules for determining the nature and extent of natural variability.

There is a risk of placing variability criteria first in the decision-analysis process. Given the wide range of natural variability and uncertainty in ecological systems, one could legitimately ask whether an anthropogenically induced change in an ecological endpoint would ever exceed natural variability. Certainly the variability of commercial fisheries data would fall into this category. Nevertheless, the question must be asked, and the answer must reflect best professional judgment. A

default position is to assume that exceedence of natural variability is the norm rather than the exception; then, based on this assumption, one should proceed through the ESF, reassessing variability only if it is deemed to be the critical component in the decision. Using variability to circumvent or delay the significance analysis process is not acceptable or recommended.

5.4.2. *Reversibility*

Reversibility is a primary topic of consideration for the risk manager when interpreting the ecological significance of the findings of an ecological risk assessment. When reviewing the results of the risk assessment, the risk manager must ask whether the types of risks and their severity are reversible if management action is taken to mitigate effects. To make an informed decision regarding reversibility, the risk manager must consider the following topics.

5.4.2.1. Nature of the Stressor

The risk manager must assess the nature (e.g., physical, chemical, biological) and properties (e.g., duration, intensity, frequency) of the stressor that are relevant to recovery. For example, the potential recovery of an ecosystem from a labile chemical stressor will be different from recovery from a cyclical biological stressor (e.g., the gypsy moth) or from the physical alteration of habitat for which reversibility may be impossible.

5.4.2.2. Ecological Organization

The risk manager must assess the structure and complexity of the ecological system or component at risk, because these factors will influence both the degree and rate of recovery. For example, the time needed for a fisheries stock to recover might be a decade or more; the recovery of a benthic infaunal community could require 3 to 5 years; a planktonic community can completely recover within weeks to months; and reforestation often can take several decades. The common ecological considerations in these examples concern the temporal scale of the systems's life history and the interspecific and trophic dynamics of the populations comprising the ecosystem or landscape ecology that is potentially at risk.

5.4.3. *Scale*

5.4.3.1. Spatial Scale

There are at least four spatial factors that need to be considered in regard to ecological significance: absolute area (km^2); the percentage of the landscape; the extent of landscape fragmentation; and the role or use of the area at risk within the landscape.

Although the absolute areal extent of the ecosystem at risk is the first spatial issue addressed in tier 1 of the ESF, a small spatial impact should not be summarily disregarded. Tier 2 addresses the roles or uses of areas within a landscape. The first consideration is whether there are critical use or habitat designations for certain portions of the landscape. For example, if the landscape includes a small area of protected wetland, this designation may be significant even though the wetland is small both in absolute area and in percentage of the total landscape. A second consideration is the percentage of the landscape that is at risk and how it relates to the territorial range and refugia of the populations and communities of concern. For example, is there an adequate stock available within the landscape for recruitment, or are there satisfactory linkages between one or more landscapes that provides refugia for affected populations such as keystone or endangered species? Are there adequate corridors, terrestrial or aquatic, for successful migration? Often overlooked in aquatic systems is the secondary risk to migratory species (e.g., anadromous fish, mammals) that can be posed by acceptable uses of certain portions of water bodies for specific times of the year.

5.4.3.2. Ecological Scale and Complexity

Ecological scale can be an important factor in determining the significance of changes in ecological assessment endpoints. Specifically, a determination should be made about whether a systems perspective can be used to interpret the significance of changes observed in measurement and assessment endpoints. For example, the significance of a large change in an endpoint that represents a highly redundant ecosystem function may be considerably less important than a small change in the population density of a keystone species. These relationships (i.e., the connectiveness of the measurement and assessment endpoints to the system as a whole) could be illustrated using network diagrams that clearly describe the importance and value of specific pathways.

5.4.4. *Magnitude (Severity)*

The magnitude of change in the endpoint is also a factor in assessing significance. Magnitude, as used in this discussion, is defined as the amount of exceedence above the natural variance. For example, Is the change in the assessment endpoint 10, 50, or 250 percent above the upper variance limit? In actuality, magnitude may be misleading, and severity may be a more appropriate term, since the concept is intended to provide a sense of the importance of the change in the assessment endpoint.

6. HYPOTHETICAL CASE STUDY

6.1. Application of the Ecological Significance Framework

6.1.1. *The Problem Setting*

A tanker carrying #2 fuel oil collides with a freighter approximately 3 km offshore. By the time emergency response, containment, and cleanup crews reach the scene, more than half the cargo—15 million barrels of oil—has been released into ocean waters. A slick approximately 2.5 km long and 1.5 km wide has developed. The water temperature is 18 °C, the salinity 33 g/kg, and the prevailing winds are onshore at 18 to 24 knots. Currents are along shore and modified by diurnal tides of 3.5 m. The coastline is irregular, with numerous small embayments that support an extensive shallow, sloping subtidal community and a rich intertidal community. Predictions from oil-spill models indicate that the oil will move toward and onto shore within the next 48 to 72 hours.

6.1.2. *Problem Formulation*

Upon examination of preliminary information, the emergency response team proposes two remedial options. Option 1 is to use a dispersant to break up the slick, which is now over 2.5 km long and 1.5 km wide and continuing to grow. While this would reduce the likelihood of extensive onshore contamination, it likely would cause extensive mortality to the phytoplankton, zooplankton, and icthyoplankton, which are important for commercial fisheries, that are in the water column at this time of year. Option 2 is to try to contain and pump off as much oil as possible; this option anticipates that a shift in wind direction will move the spill away from shore and allow for natural dispersal at sea. If this does not happen, the oil will contaminate the extensive sub- and intertidal mud flats, the rocky intertidal communities, and beaches and pose an additional hazard to avian and mammalian fauna.

There are clear risks to community structure and function with either option. Thus, it is assumed there will be a demonstrable change beyond natural variability in the ecological assessment endpoints (e.g., integrity of planktonic, benthic, and intertidal ecosystems), each of which is valued by society. Having defined the risks for both options, the emergency response team next must determine the ecological significance of each of the risks resulting from the two remedial options.

6.1.3. *Application of Significance Criteria*

The following illustrates the application of tier 1 of the ESF (see figure 2-5a) to each of the potential risks outlined in the case study.

- ***Variability.*** Do the changes in the ecological assessment endpoints (e.g., integrity of the planktonic, benthic, and intertidal community structure and function) exceed natural variability? Planktonic populations often are characterized by extensive spatial and temporal variability. Nevertheless, within the spatial boundaries of the spill, the use of dispersants is likely to produce complete mortality of all planktonic forms within the upper 3 meters of water. Benthic and intertidal communities generally are stable and have less spatial and temporal variability than planktonic forms. Under both remedial options, it is likely that the changes in the ecological endpoints will exceed the natural variability for threatened communities in both space and time.
- ***Magnitude.*** The magnitude of change in the endpoints is likely to vary but in general be quite severe. Treatment with dispersants is projected to be lethal to all types of plankton in the upper 3 meters of water, and it is likely that contamination of the benthos and intertidal habitats will result in severe impacts on survival and likely chronic effects lasting for several years. It might be difficult to distinguish the severity of effects between the two options.
- ***Spatial Scale.*** The areal extent of impacts is similar for each of these options. The slick is 6 km^2 and is projected to contaminate approximately 4.5 km of coastline if permitted to go ashore. In both cases, while extensive, the area of impact constitutes a small percentage of the landscape. This leaves considerable area available for replacement stocks and creates significant fragmentation of either the planktonic or inter- and subtidal habitats. Since this area is not a mammalian or avian migratory corridor, this is not an ecologically significant problem.
- ***Time of Reversibility.*** The implicit assumption in this analysis is that the effects to the ecological endpoints (e.g., populations, community, and ecosystem level effects) are reversible over some time period. Further, once recovered, the structure and function of the respective ecosystems will be similar if not identical to those that preceded the spill. Given these assumptions, the issue of significance centers on the time of reversibility (i.e., not whether recovery will happen but when).

The time needed for reversibility of changes in phytoplankton and zooplankton populations should be short (days to weeks) given their rapid generation times and easy immigration from adjacent water

masses. In contrast, since ichthyoplankton do not reproduce, the effects on this community would in a sense be irreversible. However, two factors need to be considered: (1) ichthyoplankton typically experience extensive natural mortality, and (2) immigration is readily available from surrounding water masses. Taken together, these factors might render the population-level losses indistinguishable from natural variability after several days.

The time needed for reversibility of changes in benthic and intertidal communities is likely to be long (years to decades). First, the stressor (oil) would be likely to persist in sediments and on rocks for several months to years. Second, the life histories of the species comprising these communities span 3 to 5 years. Third, the reestablishment of benthic intertidal community and ecosystem structure (hierarchical composition and function) often requires decades.

Ecological significance. Both options result in (1) effects to the ecological endpoint that are of great magnitude, (2) exceedences of any reasonable estimates of variance for those endpoints, and (3) similarities in estimates of areal impact. Thus, what distinguishes the significance of risks between the options is not magnitude/severity or spatial scale of effects, but the time of reversibility and system recovery. The significance of the ecological changes to the benthic and intertidal ecosystems is considerably greater than to the plankton. The remedial option of choice would be to disperse the oil, effectively preventing it from reaching shore where it would contaminate the benthic and intertidal communities.

7. ECOLOGICAL SIGNIFICANCE INPUTS TO DECISION MAKING

7.1. Weight of Evidence in Decision Making

Although a discussion of ecological risk management and risk communication is beyond the scope of this chapter, this section briefly addresses certain aspects of these areas as points of connectivity within the overall process.

Defining ecological significance involves addressing ecological issues concerning the importance of change in regard to one or more ecological endpoints and societal issues concerning the explicit or implicit importance of the ecological system in its altered state. Because there are no simple formulas to follow, in our view the decision-making process involving risk management and communication should rely on a weight-of-evidence approach. In this way, the judgment of the decision maker, supported by appropriate ecological and social science expertise, can be made on a range of factors.

The first aspect to consider is the specific finding that an anthropogenic change is ecologically significant. Ultimately, this must be determined by the risk manager and decision maker, not by the scientists alone. All ecological, societal, and values issues should be factored into the determination, along with such considerations as intensity of the stress, magnitude and spatial extent of the response, time for ecological recovery, importance to society of the changed components of the ecological system, and economic and other societal costs of management options.

Consider, for example, the ecological and agricultural consequences of global climate change. If this possible anthropogenic stress is realized, the effects would be experienced by a broadly diverse array of plants and animals in virtually all types of habitats, at spatial scales from minute to global, and at temporal scales from days to centuries. Thus there is no simple database or single model that can be used to define likely ecological consequences; rather, a suite of appropriate methodologies would have to be compiled, including, for instance:

- Historical analogs of extreme climate events and their consequences (e.g., effects of the heat and drought in the summer of 1988 on agriculture of the United States);
- Statistical models, also based on historical records but with an emphasis on the distribution of stress-response correlations rather than extreme events only;
- Physiological databases (e.g., from laboratory experiments on the temperature or soil moisture tolerances of different individuals from various test species);
- Whole-ecosystem manipulations (e.g., use of microcosms to examine community and ecosystem dynamics in response to changes in the physical environment);

- Paleo-analog records relating the distribution of plant and animal species under various climatic conditions;
- Life-zone classifications that relate biome distributions with physical conditions;
- Process-based ecological models; and
- Expert judgment.

To assess adequately the potential consequences of global climate change on ecological systems from population to biome levels, the full suite of such methodologies would be needed, since any single methodology has limitations and inherent uncertainties. The array of methodologies also would allow for the derivation of conclusions without the constraints imposed by individual approaches. The diversity of databases and analyses would be incorporated into the weight-of-evidence determination. Similar multiple-methodology approaches would be appropriate for many other anthropogenic stresses, especially for those involving nonchemical stresses, multiple stresses or multiple target ecological systems, and incomplete databases.

7.2. Decision Making in the Presence of Uncertainties

In determining ecological significance and weighing this with economic, societal, or other factors in risk management, the decision maker must accommodate uncertainties. For any environmental problem facing society today, there are uncertainties concerning, for example, how ecological systems function and respond to stress and how recovery processes operate; the availability of data at the various scales of interest; requirements for extrapolating from laboratory or analog data to real-world ecosystems; and the continual presence of natural variability in both the physicochemical environment and in biological organisms and processes. Uncertainty will always be a factor in risk assessment, regardless of how much research is conducted or how extensive a database is established. Thus environmental decisions must be made despite the uncertainties, otherwise they would literally never be made. Using the weight-of-evidence approach, a 95-percent confidence level is not necessary or in many cases even feasible. Rather, the risk manager and the decision maker must make determinations based on information from analytical techniques and available databases.

7.3. Adaptive Management

Along with relying on multiple analytical tools and assessments, decision making in the presence of uncertainties can be facilitated by using adaptive management. This technique calls for:

- Establishing appropriate monitoring for ecological endpoints judged to be responsive to management decisions;
- Following the state and trends regarding endpoints and evaluating new information in the context of anticipated responses or control conditions; and
- Modifying management decisions based on evidence of the success, failure, or other more subtle indicators of the efficacy of the management policy.

Adaptive management reduces the need for certainty, allows the adjustment of policies as societal values change, and accommodates natural variability in important ecological endpoints (Holling, 1978; Walters, 1986). Refining interim decisions after careful monitoring of endpoints can permit early reduction or mitigation of stress and can limit the consequences of a misjudgment.

There are limitations to this approach relating particularly to time lags regarding responses (e.g., controls on the emission of chlorofluorocarbons may take several decades to be manifested in reduced stratospheric ozone concentrations) or relating to societal and institutional factors (e.g., public and political perceptions that an environmental problem is "solved" and thus no longer of concern). Nevertheless, given the uncertainties and complexities, ecological risk management and environmental decision making must be adaptive if they are to be successful. This concept has been incorporated into the risk assessment framework in the form of feedback loops that lead from decisions back to the problem formulation and analysis steps. Nonetheless, as the guidelines are further developed, the adaptive management concept should be incorporated more extensively.

7.4. Research in Support of Decision Making

Clearly the focus of environmental research should be the minimization of uncertainties and the reduction of risk. Thus two basic types of research are needed:

- Development and improvement of assessment tools (e.g., new models of ecological systems or experimental manipulations on more complex and larger-scale systems); and
- Expansion of our understanding of stress-response relationships and the consequences of anthropogenic activities (e.g., improved dose-response databases on chemical mixtures affecting communities, or improved physiologically based models to describe the biochemical, individual, and pollution-level responses of a particular species).

Also, a great need exists for research on the recovery and ecosystem-shift aspects of stress ecology (i.e., on the rates and completeness of recovery once an anthropogenic stress is reduced or

eliminated). Similarly, research is needed on mitigation actions a risk manager could implement that might alter the effects of a stressor.

7.5. Applicability of Ecosystem Management Principles

7.5.1. *Ecosystem Management Principles*

Ecosystem management is emerging as an innovative framework for achieving the harmonious and mutually dependent sustainability of society and the environment (U.S. MAB, 1994a). The principles of ecosystem management, which focus on human and natural systems at regional scales across intergenerational time periods, are being developed by a variety of groups, including the U.S. Man and the Biosphere Program (U.S. MAB, 1994b). In general, the principles, which must be considered as an integrated whole, include the following:

- Use an ecological approach that is focused on recovering or maintaining the biological diversity, ecological functioning, and defining characteristics of natural ecosystems.
- Recognize that (1) humans are part of ecosystems and that they shape and are shaped by natural systems, and (2) the sustainability of ecological and societal systems are mutually dependent.
- Adopt a management approach that recognizes ecosystems and institutions as characteristically heterogeneous in time and space.
- Integrate sustained economic and community activity into the management of ecosystems.
- Develop a shared vision of desired human/environmental conditions.
- Provide for ecosystem governance at appropriate ecological and institutional scales.
- Use adaptive management as the mechanism for achieving both desired outcomes and new understanding regarding ecosystem conditions.
- Integrate the best available science into ecological decision-making processes, while continuing scientific research to reduce uncertainties.
- Implement ecosystem management principles through coordinated government and nongovernment plans and activities.

7.5.2. *Applicability of Ecosystem Management to Ecological Significance*

Adopting these management principles for ecological risk assessment has considerable implications for ecological significance, since they concern many of the relevant ecological/societal value determinations. For example, identifying areas to be protected from human-induced alterations

means that the threshold for the significance of ecological effects in that area would be much lower than in another part of the landscape. Thus establishing distinct ecological goals for different parts of the landscape defines ecological significance.

Consider the case of South Florida, where extensive modifications to the surface hydrology have caused major alterations of the Everglades ecosystem (Davis and Ogden, 1994). Some parts of that landscape and seascape are protected by society as conservation areas (e.g., Biscayne National Park, Everglades National Park, Big Cypress National Preserve). Consequently, an anthropogenic stress with an adverse impact on a protected part of the regional ecosystem (e.g., nutrient additions or an altered hydroperiod) would constitute ecological significance. The same stress on a portion of the landscape designated for urbanization would not be considered ecologically significant.

Another important aspect of ecosystem management principles is the explicit consideration of spatial (e.g., regional) and temporal (e.g., intergenerational) scales (see section 4). A third aspect concerns an adaptive approach to decision making (i.e., altering the significance determination as new information becomes available).

The full implications of ecosystem management principles as direct guidance for determining ecological significance are beyond the scope of the discussion. We believe, however, that the further integration of ecological risk assessment and ecosystem management paradigms promises better environmental decision making. Such improvements will be necessary for long-term environmental protection and sustainability as human populations and activities advance into the next century.

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Issue Paper
on

CONCEPTUAL MODEL DEVELOPMENT

Lawrence Barnthouse
Environmental Sciences Division
Oak Ridge National Laboratory
Oak Ridge, Tennessee

Joel Brown
Department of Biological Sciences
University of Illinois at Chicago
Chicago, Illinois

Prepared for:

Risk Assessment Forum
U.S. Environmental Protection Agency

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1. INTRODUCTION

Conceptual model development is the final step in the problem formulation phase of the U.S. Environmental Protection Agency's (EPA's) *Framework for Ecological Risk Assessment* (U.S. EPA, 1992). For purposes of discussion in this chapter, we define a conceptual model to be a schematic summary of the planned assessment approach that (1) describes how a given stressor might affect the ecological components in the environment, (2) describes the relationships among the assessment and measurement endpoints, the data required, and the methodologies that will be used to analyze the data, and (3) summarizes the steps that will be taken to ensure that laboratory or field data collected for the assessment will be sufficient to achieve the intended objectives. In short, development of the conceptual model serves as a focusing process for conducting the analysis phase of the assessment and for defining the types and quantity of information available for risk characterization.

Although the term conceptual model development may be new to ecological assessment, the idea itself has been a part of environmental impact assessment at least since the mid-1970s. Sanders et al. (1978), in describing approaches to development of monitoring programs designed to measure ecological impacts of new energy production facilities, defined a phase called "prediction" that involved (1) synthesis of existing information about the characteristics of the facility, composition and toxicity of expected effluents, and characteristics of potentially exposed ecosystems; (2) synthesis of a box-and-arrow diagram of causal relationships within the ecosystem; and (3) generation of a set of "impact hypotheses" that would guide the collection and interpretation of subsequently collected data. Beanlands and Duinker (1984) defined a "framework" for project-specific environmental impact assessments that includes description of causal linkages between project activities and valued ecosystem components and development of explicit impact hypotheses. Westman (1985) described a five-phase impact assessment sequence in which the first two phases correspond closely to the problem formulation phase of the EPA framework and culminate in the identification of specific ecological impacts to be addressed in subsequent experimental, field, and modeling studies.

Both earlier and more recent discussions of problem formulation have emphasized that successful completion of a risk assessment requires the development of a logical analytical plan that identifies the most likely consequences of the action being contemplated and defines the data and analytical methodologies required to resolve existing uncertainties. Within the EPA framework, the conceptual model performs this critical role. The steps in formulation of the problem, culminating in the development of the conceptual model, include:

1. Classifying the problem in terms of the management context, the ecological context, and the temporal, spatial, and regulatory constraints;
2. Summarizing available data concerning the stressors and the ecosystems at risk;
3. Determining assessment and measurement endpoints; and
4. Assembling the conceptual model: causal pathways, impact or recovery hypotheses, model/data selection criteria, and measurements or models to be used in the analysis phase.

Step 1 is primarily policy driven and corresponds more or less directly to the phase in the framework concerning discussion between the risk assessor and the risk manager (i.e., planning) (U.S. EPA, 1992, see page 9). The objective of this step is to define the management bounds of the assessment in terms of the kind of decision to be made, the possible management actions, and the kind of information required to support the decision. Step 2 is an information acquisition step in which existing physical, chemical, and biological data relevant to the assessment are identified. Data sources must be evaluated to determine whether available information is sufficient or whether additional data are needed. Step 3 involves summarizing the problem in the form of assessment and measurement endpoints (Suter, 1989). These are formalized ecological equivalents of the standardized cancer risk endpoints (e.g., 10^{-6} lifetime risk) and data types (e.g., rodent bioassay) used in human health risk assessment. Step 4 synthesizes all of the previous steps in a form suitable for guiding the subsequent collection and analysis of data.

Although it is expected that all risk assessments will require a conceptual model, the way in which the model is developed will vary greatly, depending on circumstances. As part of many regulatory programs (e.g., EPA pesticide and toxic chemicals programs), relatively standardized risk assessments are performed for large numbers of individual chemicals; thus some or all components of the conceptual model may be written directly into program guidance. For other situations, including most National Environmental Policy Act (NEPA) environmental impact assessments and Superfund investigations, conceptual models must be developed separately for each assessment.

The discussion of conceptual model development in this chapter presupposes that goals for environmental protection and management (e.g., what we want the ecosystem to look like, how we balance natural preservation and economic development, how much ecological change is considered to be acceptable) have already been determined (for a discussion of these value-driven issues, see chapter 2 on ecological significance). The purpose of this chapter is to discuss the translation of agreed-upon values and goals into technically credible and cost-effective risk assessments.

1.1. Case Studies

Throughout this chapter, we use examples to reveal the diverse aspects of conceptual model development. The variety of examples helps reveal the full spectrum of possible circumstances and scenarios that can arise, although each example provides an incomplete view of the development of particular models. We continually refer, however, to three specific case studies in an effort to provide continuity throughout the discussions.

1.1.1. *Special Review of Granular Carbofuran*

In the late 1980s, EPA's Office of Pesticide Programs (OPP) undertook a special review of granular formulations of the broad-spectrum insecticide/nematicide carbofuran in light of evidence that it posed a risk to birds. OPP undertook this review based on the Federal Insecticide, Fungicide, and Rodenticide Act [FIFRA], which authorizes cancellation of registration for any pesticide that poses an unreasonable risk to humans or the environment. The special review process used data from laboratory toxicity studies, field studies, and reported incidents of bird kills to assess the potential for adverse impacts to avian species. Based on this information, EPA proposed cancellation of registration for the use of granular carbofuran, concluding that it "generally poses unreasonable risks to birds through direct and secondary poisoning" (see Houseknecht, 1993).

Carbofuran (2,3-dihydro-2,2-dimethyl-7-benzofuranyl methylcarbamate) is applied in granular form, through a variety of techniques (i.e., band, in-furrow, and aerial), to 27 different types of crops for the control of certain insects and nematodes. Following application at the beginning of the growing season (e.g., April to June), either the pesticide becomes incorporated in the soil, where degradation occurs fairly rapidly, or the granules remain intact on the soil surface, where the pesticide may persist for over 60 days. Birds may ingest the granules accidentally during normal ground foraging, insectivorous birds may ingest exposed invertebrates, and scavenging or predatory birds may ingest exposed birds and other vertebrates.

Suspicions about adverse effects to birds from carbofuran arose from reports about 40 separate die-off events claiming from 1 to more than 2,000 birds. The diversity of poisoned birds (i.e., waterfowl, robins, songbirds, shorebirds, and raptors) indicated several direct and indirect avenues of exposure. Exposure studies found carbofuran granules and residue on the soil surface, within earthworms, and within bird carcasses. In addition, other information linked carbofuran with adverse effects in birds: (1) carbofuran is acutely toxic to birds—in some cases a single granule can kill a small bird; (2) whole-body residues of carbofuran within bird carcasses from purported die-offs

frequently have exceeded LD_{50}s; and (3) systematic searches of fields following carbofuran applications have found bird mortality rates of at least 0.1 to 3.6 birds per acre (Houseknecht, 1993).

In this case study, avifauna is the component of the environment under protection, with particular attention given to hawks, eagles, and owls. The regulatory endpoint concerns the decision of whether to revoke the registration of granular carbofuran.

1.1.2. *Modeling Future Losses of Louisiana Forest Wetlands*

The Lake Verret Basin, Atchafalaya floodplain of the Mississippi River delta, in Louisiana, includes a variety of bottomland forest types, agricultural fields, industrial parks, urban areas, and a system of levees dating from the 1930s. Activities such as petroleum pumping, land development, dredging, and shipping may have increased rates of erosion and silt runoff, while the levees may have reduced the rates of siltation and sedimentation. Increased erosion and reduced sedimentation could increase subsidence rates and expose the bottomland forests to increased flooding, elevated water tables, and increased salinity. Moreover, subsidence could promote a successional process in which the community composition of 16 principal tree species shifts from dry to wet bottomland hardwood, then to swamp (cypress-tupelo forest), and eventually to unforested marsh or open water. Such changes in forest structure can have concomitant effects on a number of bird species (e.g., the downy woodpecker, the wood duck) and species of mammal (e.g., the gray squirrel, swamp rabbit, mink) that live within these different habitats.

With wetlands protection as the goal,[1] EPA's Office of Policy, Planning, and Evaluation (OPPE) undertook this case study and focused on predicting changes in the wetlands of the Lake Verret Basin. To predict changes in forest composition under actual and hypothetical rates of subsidence and changes in two bird and three mammal species, the risk assessment parametrized a bottomland succession model (FORFLO [Pearlstine et al., 1985]) and used the model's output as input for the habitat suitability indices (HSI). The forest model requires data on the existing vegetation (e.g., relative abundances, size, densities of canopy tree species), hydrologic data (e.g., average water heights and water-table depths over 24 half-month periods), and site data (e.g., elevation and soil

[1]"In 1987, at the request of the EPA Administrator, a National Wetlands Policy Forum convened to suggest ways to improve wetland regulation and management. In its final report, *Protecting America's Wetlands*, the Forum recommended '...no overall net loss of the nation's remaining wetlands base, as defined by acreage and function....' At present, EPA lacks risk assessment and management approaches for considering physical habitat alteration and biological diversity." (See Brody et al., 1993.)

type). Once parametrized, FORFLO projected future changes in the abundance, size, and density of the 16 tree species. For instance, at present rates of subsidence, the wet bottomland hardwoods will be replaced completely by cypress-tupelo swamp within 50 years and the succession of dry bottomland hardwoods to swamp will be complete within 120 years (Brody et al., 1993).

For this case study, the valued components of the environment included the dry bottomland hardwood forest, the wet bottomland hardwood forest, and the cypress-tupelo swamp. Changes in subsidence rates constitute the stressor, the sources of which include earlier levee construction on the Atchafalaya and Mississippi rivers. Despite a regulatory climate that encourages wetland protection, no person, agency, or entity capable of making a management decision based on the risk assessment was available. As a result, the risk assessment provided only a means for making projections based on present conditions in regard to assumptions about a variety of land-use practices and subsidence rates.

1.1.3. *Assessment of a Freshwater Stream and Wetlands Near an Inactive Coke Production Plant*

This case study describes a Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) assessment of an inactive coke production facility in southeastern Ohio (Davis and Bascietto, 1993). The assessment was performed before the development of any formal guidance on CERCLA ecological risk assessments and consequently is incomplete in some respects. This case is instructive nonetheless because the contamination scenario addressed is similar to those at many CERCLA sites and the studies performed to support it are similar to those that would be performed for such a site today.

The Ohio facility operated from about 1920 to the early 1980s, producing crude tar, coke, light oil, and ammonia. Until the late 1960s, wastewater and solid wastes were discharged into wetlands east of the plant, adjacent to a tributary of the Ohio River. The waste streams included process wastewater, coke and coke fines, decanter tank car sludge, boiler ash, and weak ammonia liquor. Because of contamination present in wetlands and lagoons on and adjacent to the site, it was declared eligible for cleanup under the National Contingency Plan (NCP), the regulation that enables EPA to undertake a remediation effort.

Remedial actions taken under the NCP are expected to be "protective of environmental organisms and ecosystems." Federal and state laws and regulations, termed applicable or relevant and appropriate requirements (ARARs), often are relied on when determining whether remedial action is needed to protect organisms and ecosystems. In the case of the inactive coke production facility, the

Clean Water Act, which requires the maintenance and restoration of the chemical, physical, and biological integrity of the nation's waters, provided ARARs that supported implementation of the NCP effort. The assessment examined the impacts of contaminants from the coking plant on potentially affected aquatic communities.

Under the NCP, EPA follows a three-step procedure: (1) evaluating and scoring the potential health and ecological hazards associated with the site, using a methodology known as the Hazard Ranking System (HRS); (2) conducting a detailed field investigation, known as a remedial investigation (RI), to confirm the need for remedial action; and (3) conducting a feasibility study (FS) to determine the most cost-effective cleanup approach.

For the coke production facility (Davis and Bascietto, 1993), the obvious source is aqueous and solid wastes released by the plant. Qualitative information concerning the stressors was obtained by applying EPA's HRS, which uses historical records and preliminary site inspections to determine whether humans or the flora and fauna in the area are exposed to hazardous levels of toxic chemicals or could be in the future.

2. PROBLEM CLASSIFICATION

The technical approach taken in risk assessment must be selected based on the regulatory/management context. For example, the Toxic Substances Control Act (TSCA) requires EPA to act on premanufacture notifications (PMNs) for new chemicals within 90 days, based on relatively little information about the structure and activity of the particular chemical. In contrast, registration of pesticides can require extensive laboratory and field toxicity testing but limited field investigation. Similarly, Superfund remedial investigations can take several years and produce substantial quantities of field data, although data are gathered under relatively uncontrolled conditions. Because these situations involve different types of scientific uncertainties, each requires a different kind of technical guidance.

Concerning determinations about the objectives and technical boundaries of a risk assessment, communication between the risk manager and the technical risk assessor is essential. The *Framework Report* implies that this communication is conducted with a single risk manager; however, these determinations can be quite complex, requiring input from many "risk managers." In some regulatory programs, especially programs involving regulation of pollutant discharge or chemical/pesticide manufacture, the problem classification step is implicit in the regulations governing the program and therefore does not need to be performed individually for each risk assessment. For such programs, time, resource, and problem area constraints are written into program-level risk assessment guidelines. In the best cases, moreover, applicants know in advance what information they must provide to risk assessors, just as risk assessors know how they will analyze and interpret the data. In other programs, especially those dealing with toxic waste disposal and environmental restoration, even determinations about which law applies may require agreement between several different Federal and State regulatory agencies.

The first, and ultimately perhaps the most important, discussion between risk managers and technical risk assessors should occur when program-level risk assessment guidelines are developed. These guidelines should reflect a consensus on criteria for establishing consistency and credibility applicable to all risk assessments performed for the particular office. The two types of guidelines relevant to the risk assessment process concern technical content and procedural criteria. Guidance on technical content consists of identification of the specific models and data types that should be used as well as the decision rules that must be followed for all risk assessments performed for the particular office. For example, OPP's standard evaluation procedures (Urban and Cook, 1986) specify (1) the kinds of toxicity test data to be used to summarize the effects on nontarget biota of a pesticide

proposed for FIFRA registration, (2) the methods for quantifying ambient environmental exposures, and (3) a set of "risk criteria" defining the action OPP may take (i.e., registration, denial of registration, or seek more data) based on different combinations of exposure and effects data.

Procedural guidance consists of general data-collection criteria that leave specifics about particular data and methods to be developed on a case-by-case basis. For example, the Office of Solid Waste and Emergency Response's (OSWER's) Environmental Evaluation Manual (U.S. EPA, 1989) provides general procedural guidelines for designing ecological risk assessments at Superfund sites without identifying what types of data to use. Such specifics are left to the development of "Sampling and Analysis Plans" and "Data Quality Objectives."

The particular office's specificity requirements for risk assessments should strongly influence the balance between technical and procedural aspects of its risk assessment guidelines. For example, because the Office of Prevention, Pesticides, and Toxic Substances (OPPTS) performs standardized assessments for large numbers of chemical substances, detailed program-level technical guidance has been developed covering exposure routes to be investigated, data to be collected, and analytical methodologies to be used (Urban and Cook, 1986; Zeeman and Gilford, 1993). In contrast, guidance for Superfund ecological risk assessments emphasizes process over technical content, since considerable guidance on sample collection, data quality assurance, and data analysis is required. For such assessments, critical issues concern data quality objectives, methods for measuring specific variables (e.g., chemistry, benthic grab samples), and statistical methodologies. Because the specific pathways and measurements as well as the appropriateness of particular models vary greatly among sites, specific guidance for certain aspects of the assessment cannot be provided in advance.

The appropriate procedures and technical content for a risk assessment should be reflected in the conceptual model. For general assessments, a conceptual model that is very simple in structure may be sufficient, and even explicitly called for in program guidance. For site-specific assessments, it may be necessary to develop a sophisticated conceptual model that includes many exposure pathways, species, and measurement approaches. Thus each model will be unique to the particular assessment, with the empirical foundation of each established by taking into account the available information on the stress regime and the ecosystems at risk.

3. STRESSOR CHARACTERISTICS

After classifying the problem, the next step in developing the conceptual model involves summarizing available information on stressors and gathering data on ecosystems potentially at risk. The conceptual model will be based on knowledge as well as hypotheses about the stressors and ecosystem components and the potential interplay between the two. Hypotheses regarding this interplay serve two essential purposes: (1) they reduce what may be a considerable number of candidate stressors and ecosystem components to a manageable few, and (2) they suggest flow diagrams linking potential stressors to ecosystem components (see section 6.1). The network of causal pathways linking stressors to various effects on measurement and assessment endpoints will ultimately constitute the conceptual model (see section 6.2).

A stressor is a chemical, physical, or biological factor that results in adverse effects to ecosystem components or functions. The stressor's characteristics represent the totality of stress- related properties that include the type and source of the stressor, the stressor's pattern of exposure, and the spatial scale at which the stressor operates. Important information includes the source; aspects such as intensity, frequency, duration, timing, scale, and mode of action; and whether the agent or stressor is chemical, physical, or biological (U.S. EPA, 1992, page 11). Information on a stressor's source answers the question, "Where does it come from?" A stressor's frequency, duration, timing, and scale answer the question concerning co-occurrence, "Where is the stressor?" And, a stressor's intensity, mode of action, and type (i.e., chemical, physical, or biological) answer the question concerning ecological effects, "What might it do?"

3.1. Source

The process of gathering information on the possible source of a stressor begins the dialogue between the risk manager and assessor regarding stressor characteristics. Identifying the source may lead the risk assessor to the regulatory endpoints and to the risk manager, who generally regulates sources rather than stressors. Moreover, most regulations differentiate among various sources that may involve the same or different stressors (e.g., sport and commercial fisheries).

Identifying the source often will allow the risk assessor to determine the scale, duration, and frequency of the stressor; its potential co-occurrence with particular ecosystem components; and the likelihood that additional stressors are related to the same or a different source. A source can be natural (e.g., tar pits) or anthropogenic (e.g., oil spills), geographically well defined (e.g., point-source pollution), or geographically vague and diffuse (e.g., automobile emissions).

In the case of granular carbofuran (Houseknecht, 1993), the source is the group of farmers who apply the insecticide/nematicide to 27 types of agricultural crops using a variety of application techniques. The source can be identified more specifically as relating to the particular form of application (i.e., band, in-furrow, and aerial) and the particular crop species. Application technique is significant because it determines the level of frequency for exposed granules (i.e., highest for aerial and lowest for in-furrow application). The significance of crop species in regard to source is that it determines the timing, dose, and circumstances of application (e.g., application of carbofuran to rice fields increases the likelihood of exposure to waterfowl and shorebirds).

The Louisiana forests wetlands example (Brody et al., 1993) shows how the risk assessor and manager may need to trace the stressor (e.g., subsidence) back to the source(s). In this case, sources of subsidence can include natural processes of erosion and compaction or a variety of human activities such as land development, dredging, generation of "propwash" from ships, recreational activities, and construction of flood-control levees. The risk assessor selected levee construction as the primary source. Natural processes were ruled out because before flood-control measures were implemented, the basin experienced a net accretion of sediments as a result of high siltation rates. Levees were implicated because they have prevented sedimentation from occurring across the basin's floodplains.

In the Ohio contaminated waste site example (Davis and Bascietto, 1993), the original source of chemical stressors was the coke production plant that stopped operating in the early 1980s. As will be the case for assessment of many contaminated sites, however, the relevant sources now are the soil and lagoons near the plant site. Thus data sampling stations were set up to monitor the effects of water runoff from the plant area. In addition, there were suspicions that untreated sewage outfall might be draining into nearby Ice Creek from areas north of the plant site.

3.2. Particular Stressor Characteristics

3.2.1. *Defining the Characteristics*

Information on the stressor's intensity, frequency, duration, timing, and scale is important for characterizing exposure (see chapter 4 on characterization of exposures), and information on the stressor's mode of action is important for characterizing ecological effects (chapter 5 on effects characterization). When the assessment is predictive, the stressors are generally known and the resulting conceptual model will be based on these stressors and their associated characteristics. The task of problem formulation then becomes one of predicting an array of ecological effects. Other

stressors enter the conceptual model only insofar as they may mitigate, compound, or add to the effects of the target stressors.

In a retrospective or ecological effects-inspired assessment, where the stressor may be unknown or hypothetical, the ecological effects serve as the basis of the conceptual model. Using the model, an array of stressors and possible sources can be predicted. Other ecological effects enter the model only insofar as their occurrence (or lack of occurrence) indicates the influence of particular stressors. In a retrospective assessment, the magnitude, scale, and type of ecological effects (and the absence of other effects) may indicate the stressor's characteristics and suggest its identity. In all cases, information about the following particular characteristics of a stressor will come from empirical, inferred, or estimated data.

3.2.1.1. Intensity

The expression "the dose makes the poison" indicates the importance of information on a stressor's intensity, because generally the relationship between the intensity of the stressor and the magnitude of an ecological effect is monotonic. Below some threshold there may be no detectable ecological effect, or the effect eventually may fall below the bounds of relevance to the risk manager. Considerations of intensity in the conceptual model comprise aspects of "dose" in exposure characterization (see chapter 4) and "dose response" in effects characterization (chapter 5).

Intensities may be measured as the concentrations or reactivities of chemical stressors, as the densities or population sizes of biological stressors, or as the magnitude of physical stressors. Chemical stressors may be elaborated further as distributions of concentrations in space or time, as estimated from solubilities, structure-activity relationships (SAR), binding properties of organic aggregates, and vapor pressures. For biological stressors, additional considerations may include the organism's invasiveness, mobility, dispersal rates, pathogenicity, or rates of predation/herbivory on other components of the biota (see chapter 6 on biological stressors). The magnitude of physical stressors can relate to the number of, for example, visitors, vehicles, hunters, or mining/logging permits associated with the exploitation of an area and the distribution of these activities in time and space. In regard to wetlands, the magnitude of habitat modification may relate to the density of building sites, the proportion of wetland subjected to dredging, or changes (or lack thereof) in the level of water tables.

3.2.1.2. Frequencies

A stressor event can be isolated, episodic, or continuous, and events can be described by the influence of periodicity (e.g., daily, lunar, seasonal) or the absence of such influences (i.e., stochastic or chaotic). In conjunction with its intensity, the frequency of a stressor constitutes an ecological disturbance (see chapter 5 on effects characterization and chapter 7 on ecological recovery).

Chemical stressors may be introduced to an ecosystem as a single event (e.g., an oil spill) but are more likely to be introduced episodically (e.g., regular applications of many agricultural pesticides) or continuously (i.e., as in the case of many pollutants). Biological stressors may occur as single events if the introduced organism (e.g., a biological control agent) dies off within a relatively short time or may be episodic if the introduced organism is self-sustaining and experiences flush-crash population dynamics. Physical stressors may be continuous, as in the case of gradual alteration of habitat, or episodic, as in the case of habitat destruction and with certain forms of exploitation by humans (e.g., those dictated by weekend recreation or seasonal hunting).

Infrequent or single-event stressor exposures focus the conceptual model on acute effects and the potential for ecological recovery (see chapter 7). Seasonal or cyclic frequencies focus the conceptual model on ecosystem components that are likely to temporally co-occur with the stressor. Continuous exposures increase the likelihood of chronic effects as well as indirect effects among ecosystem components mediated through trophic interactions among species. The Ohio contaminated site, for example, may be affected by two levels of stressor frequencies. At one frequency, toxics may be leaching slowly and continuously into Ice Creek. At another frequency, there may be pulses of toxic runoff from the plant site after heavy rains or flooding of the contaminated lagoons.

3.2.1.3. Duration

A stressor's properties as an ecological disturbance influence the duration of the stressor's effects. For many stressors, the duration of indirect effects can be longer than for direct effects. As duration increases, the focus of the conceptual model will tend to shift from acute to chronic effects and from direct ecological effects to indirect effects, such as biomagnification through a food chain. Stressor duration also has direct relevance for ecological recovery issues (see chapter 7).

For chemical stressors, important information includes the substance's persistence in water, benthic sediments, soil, and air as well as the rate and extent that the chemical will degrade under particular conditions. The duration of a biological stressor depends on whether the organism

proliferates and becomes established as a self-sustaining population. The duration of physical disturbances ranges from very short to permanent.

3.2.1.4. Timing

When the frequency of a stressor is episodic, information on the stressor's timing with regard to seasonality and other biological cycles may be important in determining ecological effects. For example, agricultural pesticides are generally applied during growing seasons, increasing the likelihood of adverse ecological effects and the number of pathways by which the stressor may enter and influence the food chain. Similarly, because most human recreational activities occur during the day, data on diurnal rather than nocturnal animals may be more relevant for the conceptual model.

3.2.1.5. Scales

A stressor may operate on spatial scales ranging from local to global and from habitat specific (e.g., the aquatic versus terrestrial component) to all habitats within an exposed area. It is important to place bounds on the spatial scale of a stressor because the more expansive the scale or the more diverse the habitats involved, the greater the number of ecosystems at risk and the larger the set of possible ecological effects. The scale of a retrospective assessment may be imposed by the observed ecological effects, whereas a predictive risk assessment may use fate diagrams or models to develop a theory about the scale. It is easier to develop a conceptual model for a site-specific assessment than for a geographically vague assessment.

3.2.1.6. Modes of Action

Information on a stressor's mode of action may be the most important characteristic for linking the stressor to the ecosystem components at risk. Characterizing the mode of action is tantamount to developing the effects characterization (chapter 5); hence, mode of action sometimes can be synonymous with ecological effects, but rarely can it be defined independent of an assessment of the potential interactions between a stressor's other characteristics and some biotic component. Indeed, in one sense the definition of a stressor's mode of action begins at the stressor's entry point or at co-occurrence with biotic processes, which initiates stimulatory or inhibitory effects that lead to the ecological consequences of interest.

In its narrow sense, definition of a stressor's mode of action should be reserved for the physiological and biochemical effects of a chemical stressor on an individual organism (e.g., narcotic,

neuroinhibitor/transmitter, hormonal substitute, nutrient). The key to mode of action, however, may be in identifying critical events that result in ecological effects. For example, while it may be useful to know that cyanide acts as a cytochrome inhibitor, it might be more useful if its mode of action is incorporated into a risk assessment as a chemical stressor that causes mortality in individual organisms. In its broadest sense, it is important to consider the stressor's modes of action at various scales. For example, the Ohio contaminated site may serve as a "killjar" for organisms such as ducks that migrate into or through the site, or it may act as a reservoir of toxics that seep into surrounding habitats or waterways. The risk posed by the site is a function not only of the chemicals' modes of action, but of the broader ecological landscape within which the site is embedded.

3.2.2. *Stressor Characteristics of Granular Carbofuran*

In the granular carbofuran study (Houseknecht, 1993), the intensity of the stressor includes the application rate (pounds of active ingredient per acre), the number of exposed granules (per square foot), the concentration in soil invertebrates (ppm), and concentration in bird carcasses available to raptors and scavengers. The granular carbofuran example illustrates that exposure frequencies are related to the characteristics of the source, properties of the stressor, and interactions of the stressor with the environment.

Application of carbofuran is episodic, occurring at the beginning of each growing season (e.g., April to June, depending on location and crop). Granules may be persistent and available to birds on the soil surface for as many as 60 days after application. Contamination of invertebrate prey and birds (i.e., secondary exposure to raptors) represents a declining and continuous source of carbofuran. As a result, stressor duration may be very short in the soil, somewhat longer on the soil surface, and longer still in invertebrates and birds.

The timing of carbofuran applications corresponds directly with breeding and migration patterns in birds, which are timed to take advantage of the invertebrate production against which the pesticide is used. The scale of carbofuran can be defined by the observations of bird die-offs (i.e., 40 incidents of carbofuran-related bird kills) and the amount of agricultural land under application (e.g., 4.5 to 5.5 million acres of corn, 0.64 to 2.0 million acres of sorghum). In birds, carbofuran acts as a neurotoxin and neurostimulant by inhibiting the action of cholinesterase. Thus the mode of action focuses the conceptual model on acute rather than chronic effects.

3.2.3. *Stressor Characteristics of the Louisiana Forest Wetlands*

In the Lake Verret Basin case study (Brody et al., 1993), the stressor's intensity could be expressed as rate of subsidence (0.5 to 1.0 cm per year), as frequency of flooding, or as the height of the water table. Flooding frequency and water-table height were selected as measures of stressor intensity because they vary seasonally, while the subsidence occurs as a gradual and continuous process. Duration of subsidence in Louisiana bottomland forest is presumed to be permanent, and the scale is defined by its present range (about 50,000 hectares of the Lake Verret Basin). The mode of action of subsidence involves increased submersion of roots in water and encroachment of salt water into the wetlands. These changing hydrological regimes alter the growth, survivorship, and establishment of the constituent tree species.

3.2.4. *Stressor Characteristics of the Inactive Coke Production Plant*

In the coking plant study (Davis and Bascietto, 1993), the intensity of the stressor is expressed in terms of concentrations of toxic chemicals in environmental media (i.e., water, soil, and sediment) contaminated by past waste releases. In Suter's (1993) terminology, this assessment is "exposure driven" because the contaminants are derived from activities that occurred in the past; the significance of those past activities must be inferred from the presence of residual exposures and effects. Stressor characterization in such situations involves identifying contaminants that can be linked to the suspected source. In this case, the contaminants are the organic and inorganic contaminants known to be associated with coke production. The stress regime is continuous and the scale (as addressed in the particular study) is limited to the plant area, the adjacent and downstream portions of Ice Creek, and the near-shore areas of the Ohio River immediately below Ice Creek. Conceivably the scale could have been enlarged to include home ranges of waterfowl or piscivorous wildlife that use the lagoons. No single mode of action can be identified for this case study because the stressor is a complex mixture of contaminants with multiple modes of action.

3.3. Types of Stressors and Stressor Characterization

Often the type of agent or stressor also can be useful in identifying regulatory endpoints, the stressor's mode of action, and likely ecological effects. This section examines the ways in which information about the three types of stressors—chemical, physical, and biological—influences the way in which the stressor characteristics are incorporated into the conceptual model.

3.3.1. *Chemical Stressors*

Chemical stressors can be divided into pesticides, hazardous contaminants, stimulatory contaminants, and radionuclides that may represent a deliberate or accidental disturbance. Such distinctions are significant because, for example, pesticides, radionuclides, and pollutants often vary in respect to applicable law, the appropriateness of the decision maker or regulatory authority, and the amount of available information and test data. It is important to be aware, however, that the same chemical (e.g., carbon dioxide) used as a pesticide to fumigate insects may turn up as a hazardous pollutant in confined spaces or as a stimulatory pollutant that increases plant growth and the efficiency of plant water use.

3.3.1.1. Pesticides

Because pesticides are in a class of chemicals that can have inhibitory effects on individual organisms, they are heavily regulated at the State and Federal levels, and considerable information about them is available. Moreover, much of the assessment process for pesticides is well defined and prescribed by existing guidelines (e.g., the *Federal Register*). Although pesticides are used to affect target populations of organisms, the likelihood that they will affect similar nontarget organisms is high. In forestry and agricultural management, applications generally are restricted to portions of confined areas to achieve specific objectives. In contrast, urban and household pesticide use may involve unlimited quantities and unrestricted applications.

An important consideration in regard to pesticides concerns scale: during and after application, many pesticides can be leached or eroded from terrestrial areas into waterways. In some uses, such as aquatic-vegetation control, pesticides may be applied directly into waterways, allowing toxics to spread downstream. Organisms also may spread pesticides by entering the affected area, ingesting or becoming coated with a toxic, and then migrating from the area. In such ways persistent and lipophilic pesticides may accumulate to extraordinarily high concentrations in soils, aquatic sediments, and top-level predators.

Given that the effects of pesticides often mimic natural agents of historically notable toxicants (e.g., alkaloids, allelopathic chemicals, antibiotics), information on their modes of action draws heavily from theories and principles of toxicology. Most insecticides, herbicides, and fungicides act directly to inhibit or stimulate enzyme action by physiologically impairing reproduction, metamorphosis, or development or by causing behaviors such as aversion. In terms of indirect

effects, pesticides can be transformed into derivative chemicals with distinct modes of action or bioaccumulate through food chains.

In a conceptual model for pesticides, the stressor characterization might first show ways in which the chemical and its derivatives may contact or enter target and nontarget biota. Then, the model will generally address dose-response relationships involving acute and chronic effects to the individual. When ecological effects include the transfer of chemicals through the food chain, an event-tree diagram may show the important aspects regarding the species' ecologies that promote or encourage bioaccumulation or spread (figure 3-1). Useful information for such a model might include acute and chronic dose-response relationships (e.g., LC_{50}, LD_{50}, EC_{50}, No Observed Effect Concentration [NOEC], Lowest Observed Effect Concentration [LOEC]). If the primary ecological effect of the chemical is to depress the abundance of one component of the ecosystem, then a food web conceptual model may show the subsequent direct and indirect ecological effects that may be transferred through the food web as a result of population interactions (figure 3-2).

3.3.1.2. Hazardous Contaminants

A much larger class of chemical stressors includes toxic by-products from commercial operations and hazardous constituents of household products. Because the introduction to the environment of such toxics may be incidental, information on intensities, frequencies, duration, and scales may be limited and variable. For example, unplanned or accidental releases of pollutants (e.g., leakage from storage tanks, failure of pollution-control devices, transportation accidents) present unpredictable events in time and space (for information on factoring stochastic events into risk characterization, see chapter 9), occur continuously or episodically, and may be of a low or high intensity. Households tend to release mixtures of toxics into the air, landfills, ground water (septic systems), and sewer systems at low intensities. In aggregate, however, household releases generally occur at large scales and with varying frequencies and durations. Although point-source pollution from industrial facilities may be of known intensity, timing, and duration, the composition of the effluent may include several unexpected chemicals as a result, for instance, of a malfunction of pollution-control equipment.

The conceptual model for such a stressor may begin with a transport, transformation, and fate diagram that postulates how the chemical(s) enters the air, water, and/or soil (see table 3-1). For chemicals under consideration, the kinetic or equilibrium constants must either be measured using standardized protocols (U.S. EPA, 1988) or estimated from empirically derived SAR or quantitative SAR. At this stage, such a diagram should provide answers to questions about where the chemicals

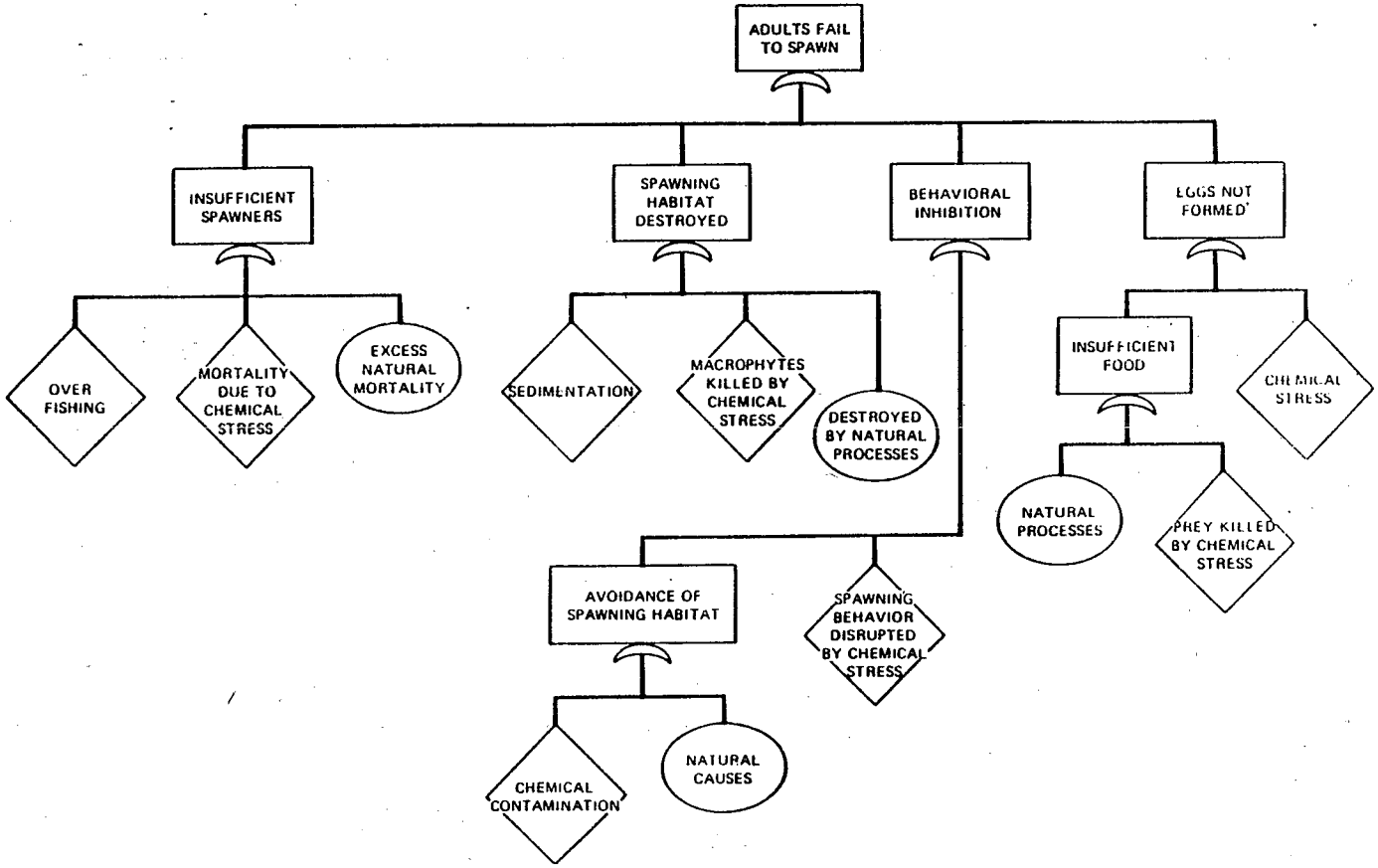

Figure 3-1. Sample event-tree diagram for illustrating the possible causes of recruitment failure in a fish population (Suter, 1993).

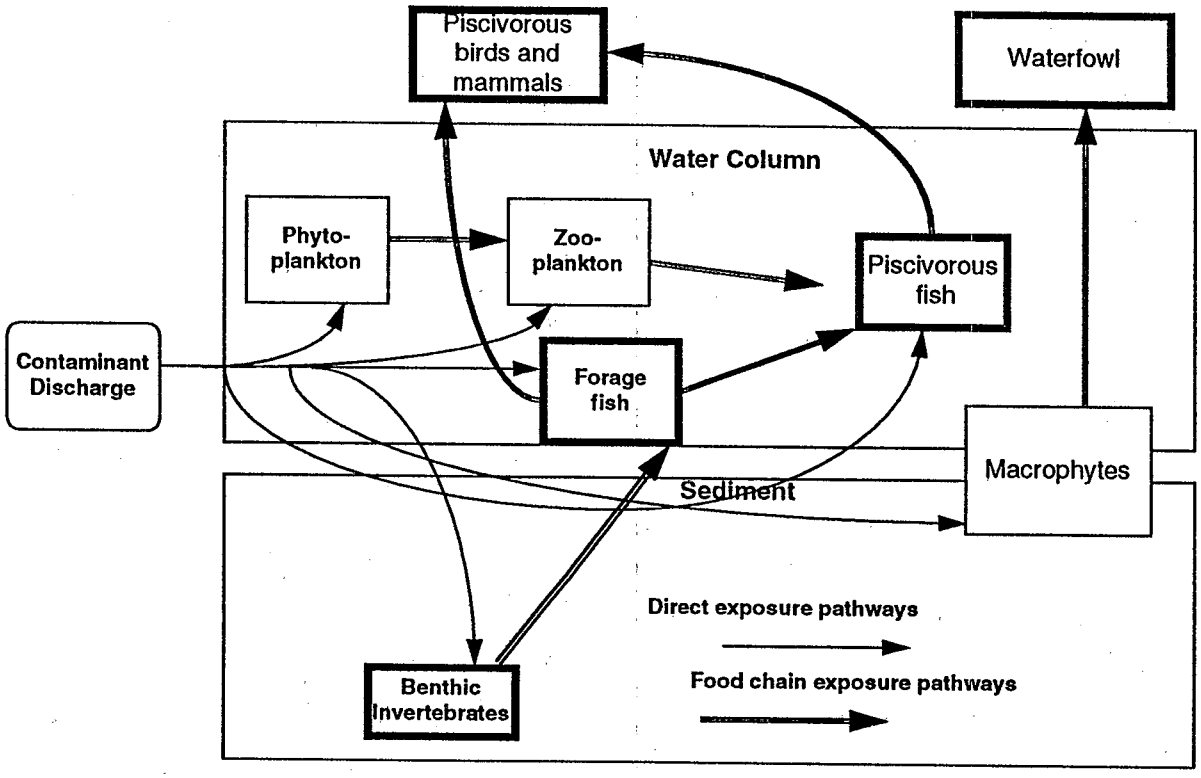

Figure 3-2. Diagram of contaminant transport pathways in an aquatic ecosystem (based on Davis and Bascietto, 1993).

Table 3-1. Selected Processes and Parameters Relevant to the Transport and Transformation of Chemical Stressors (after Mill, 1993)

Process (rate constant)	Parameter (properties of environment)
Atmosphere	
Photolysis	Light intensity
Oxidation	Oxidant concentrations
Rain out	Precipitation rate Sticking coefficient
Transport	Wind velocity
Surface waters	
Volatilization	Henry's Law constant Surface roughness
Sorption/bio-uptake	Organic/lipid content of sediments and/or organisms
Hydrolysis	pH, temperature
Photolysis	Light intensity
Biotransformations	Organism population, nutrient level, temperature, pH

Note: Processes and their associated parameters may be incorporated into the conceptual model depending on the system and the available data. The resulting transport and fate model may provide the basis for analyzing exposure (see chapter 4 on characterizing exposures).

and their derivatives will go and what their respective initial or equilibrium concentrations will be in the air, soil, water, or sediments. The distribution of the pollutant may take the form of a mass balance model (Mackay and Paterson, 1993) in which the continuous release of the pollutant may generate an equilibrium distribution within various abiotic or biotic "compartments" of the environment.

While the transport and fate model may account for some sorption and uptake by the biota, additional aspects of the chemical should be considered in terms of the biota's transformation and metabolism (e.g., biodegradation), uptake and storage (e.g., bioaccumulation and biomagnification), and elimination. Biodegradation may be a particularly important consideration regarding the duration and intensity of the stressor. Estimates of biodegradation based on laboratory cultures or chemical structure, however, may be biased and thus suspect (Klecka, 1985), especially since biodegradation may depend heavily on characteristics of the environment (e.g., pH, temperature, moisture, oxygen, substrate density and compaction, and the availability of other nutrients) (Walker, 1978).

Toxicokinetic models, which are mathematical simulation models that estimate the internal dose of toxic chemicals from estimates or measurements of the external dose (Suter, 1993), can provide predictions about uptake, absorption, or bioaccumulation pathways of chemicals from the abiotic and biotic environment. A final consideration regarding the fate and transport of chemicals concerns the potential for biomagnification through the food chain. If this potential is established, the focus of the conceptual model may be drawn toward higher trophic levels and top-level predators. As with pesticides, these toxic effects will generally be formulated as a dose-response model of chronic and acute effects.

3.3.1.3. Stimulatory Contaminants

Contaminants such as phosphate, nitrogen, and pH can alter nutrient cycles of an ecosystem, acting as fertilizers for some organisms in the biotic community. The consequences of increasing the nutrient load of a system can be profound and highly idiosyncratic (Abrams, 1993), with measurable concentrations of nutrients increasing or declining. Generally, however, certain nutrient concentrations will increase, while others decline (Tilman, 1982). As a result, the abundance of organisms that directly use a particular resource may increase or decline (Hairston et al., 1960; Oksanen et al., 1981) and the relative frequency of constituent species may change dramatically. Much of the increase in nutrient loading may be manifested at the level of the predators that prey on

organisms using the particular resource. This may lead to the paradox of enrichment; namely, feeding the prey may simply result in more predators (Rosenzweig, 1971).

When a chemical stressor acts as a fertilizer, ecological effects that should be anticipated include changes in resource concentrations, changes in densities of constituent species, and changes in community composition as some species become excluded from the community and others are able to colonize the altered community. Theories pertaining to population interactions can provide the concepts and empirical data necessary to hypothesize the direction and magnitude of change in the exposed ecosystem. Important concepts include direct effects, such as competition and predation, and indirect effects, such as resource competition, apparent competition, mutualism, and changes in behavior. Resistance (i.e., how much will community composition change?) and resilience (i.e., how quickly and reliably will a community return to its original configuration following a disturbance?) are also important issues (see chapter 7 on ecological recovery). Consideration of these concepts will yield a conceptual model capable of predicting effects on keystone species, species compositions, and ecosystem processes.

An individual or population-level dose-response model that successfully characterizes the stress regime of a pesticide or hazardous contaminant may not be appropriate for characterizing stimulatory contaminants. While population ecology should be important for other stressors, the lowest level of interest in stressors with stimulatory effects focuses on whole populations and, more often, on population interactions among species. Instead of a dose-response model, a food web-type conceptual model (see figure 3-2) or an ecosystem model based on nutrient cycling (e.g., of carbon, nitrogen, phosphorus, sulfur) may provide a better characterization. In general, the addition of a nutrient initially will benefit mostly the species whose population size is largely limited by the resource. As this population increases in number, the species may alter concentrations of other nutrients, outcompete other species for nutrients, and benefit its predator. The predator in turn may decimate relatively rare prey species as the predator population increases in number from consuming the "fertilized" prey (this negative indirect effect of two prey species on each other via a shared predator is known as apparent competition [Holt, 1977; Holt and Kotler, 1987]). Such population interactions begin a cascade of direct and indirect effects throughout the food web. As a result, indices of biotic integrity may prove more useful in the conceptual model than information on the dose-response relationships of individual species.

Eutrophication illustrates how stimulatory contaminants may best be modeled at the level of communities. Eutrophication from the addition of phosphates or nitrogen (e.g., ammonia in the

lagoons of the Ohio contaminated site) results in aquatic systems that are rich in carbon sources and severely depleted of oxygen. Aquatic invertebrates that have low oxygen demands (e.g., some midge larvae) or hemoglobin (e.g., tubifex worms) or that obtain oxygen from the atmosphere (e.g., mosquito larvae) become favored over those that have gills and relatively high oxygen demands (e.g., mayfly larvae). For instance, as eutrophication proceeds, air-breathing pulmonate snails replace gilled snails. Eutrophication in the Ohio coke plant's lagoons provides an illustration.

A number of contaminants may have both toxic and stimulatory effects, depending on concentrations, durations, and the organisms under consideration. For example, urea and carbon dioxide can directly increase primary productivity; urea also can provide a valuable nitrogen source for plants, and carbon dioxide can increase plant growth rates and water-use efficiencies, which can change community composition by favoring woody plants over forbs and grasses. Alternatively, urea can poison and reduce growth rates in secondary consumers (e.g., many vertebrates) by creating electrolyte imbalances, disrupting protein metabolism, and increasing water loss through excretion. Taking these alternative poisonous and nutritive effects for the same chemical stressor into consideration requires rather different scales of perception. The toxic effects may be formulated as a dose response relating exposure to chronic and acute effects on individuals, while the stimulatory effects may be formulated as effects on productivity, biomass, population sizes, and the composition of species within a community.

3.3.1.4. Radionuclides

The unique hazard posed by radionuclides is the release of various types of radiation that may disrupt the integrity and normal functioning of an organism's tissues (i.e., radiation sickness), result in nonviable or mutated sperm and egg cells, or increase the level of somatic mutations. Radionuclides often represent external exposure that is easy to measure accurately. Although radionuclides can be assessed using a dose-response model at the level of the individual, the radioactive ion or element also may be toxic. The combined radiation toxicity and chemical toxicity of many heavy-metal radionuclides encourages the use of a conceptual model that presents alternative hypotheses for these two different avenues of toxicity.

3.3.2. *Physical Stressors*

This section provides a more detailed discussion of physical stressors than the above discussion on chemical stressors because (1) the majority of work already done on ecological risk assessment has

focused on chemicals; (2) the Science Advisory Board (U.S. EPA, 1990a,b) has identified the need for addressing issues of species extinction, habitat alteration, and habitat destruction; (3) the types of information available and the types of information needed may differ substantially for chemicals and physical disturbances; and (4) characterization of physical stressors generally draws more heavily from a different body of theory (i.e., population ecology) than chemical stressors (i.e., ecotoxicology).

Physical stressors can be divided into exploitation, habitat change, habitat destruction, and landscape effects. These distinctions are significant because as one moves along the continuum from human activities such as recreation or resource harvest to habitat removal and development, the focus of an assessment may change from concern for the species or affected area to concern about ecological effects on adjacent or distant areas.

Natural disturbance regimes contribute to the dynamic stability of "healthy" ecosystems by promoting species diversity, resetting successional stages, and fostering nutrient cycling and productivity. Anthropogenic disturbances may either mimic natural disturbance regimes and thus result in little or no adverse ecological effects or jolt systems into novel successional trajectories that lead to community compositions deemed less valuable (see chapter 2 on ecological significance and chapter 7 on ecological recovery).

Because risk assessments of physical stressors often confront the conflicting land-use goals of exploitation and preservation, the risk assessor should know in advance what evidence of potential ecological effects will suffice to permit the risk manager to take action. Thus regulatory endpoints relevant to the risk manager (e.g., the Endangered Species Act, Wetlands Protection Act, Marine Mammals Act) must be included in the scientific process when appropriate. Four ecosystem components are readily identifiable for this purpose:

- The manager may be interested in a target species, subspecies, or ecotype (e.g., units of conservation governed by the Endangered Species Act);
- The manager may be interested in maintaining ecosystem functioning, integrity, and self-sufficiency (e.g., preserving the viability and functioning of wetlands or the watershed functioning of upland habitats);
- The manager may be interested in preserving biodiversity (e.g., restoration and preservation of prairie plant communities); and
- The regulatory environment may dictate additional constraints and considerations (e.g., recreation, zoning ordinances, multiple land and water uses).

3.3.2.1. Human Exploitation

Even a seemingly innocuous use of an area may result in adverse ecological effects. For example, tourists pursuing cheetahs on the Serengeti with their cameras prevent them from hunting for food. The mere presence of humans may represent a set of direct and indirect physical stressors (e.g., trail damage by hikers and off-road vehicles, plant collecting and animal poaching, noise pollution).

The ways in which the presence of humans may stress an ecosystem are specific to the activity, season, and affected species. Often human exploitation of an area will be concurrent with the effects of other chemical and physical stressors. Because we hesitate to label ourselves a major ecological stressor, nonhuman stressors may initially receive more attention. Moreover, human exploitation may be difficult to discern, and data on the intensity and types of human activities within an area may be unavailable, incomplete, or difficult to collect. Yet the rapid recovery of wildlife at a number of Superfund sites indicates that chemical stressors sometimes may pose less of an ecological problem for ecosystems than the presence of humans.

Besides human intrusions and the associated ecological effects, harvesting represents a commercially or recreationally valuable activity that increases the mortality of target and nontarget species alike (e.g., marine mammals and birds caught in oceanic drift nets). Thus agencies such as the U.S. Forest Service and state fish and wildlife agencies are responsible for preserving such activities to some degree. In some cases, the physical stressor of harvesting may be compounded by the introduction of non-native species (see chapter 6 on biological stressors). Conceptual models of harvesting effects can be based on stock-recruitment models for estimating the yield and population consequences of different harvesting regimes. Such models incorporate life-history data on the species' population dynamics along with information on spatial and temporal uncertainties, density dependence (e.g., compensatory mortality), and population interactions (e.g., competition from nontarget species). These models are widely used to set harvesting quotas.

3.3.2.2. Habitat Change

Even without the influence of anthropogenic stress, ecosystems tend to be in a state of change. The dynamic nature of ecosystems is attributable to both biotic factors (e.g., population interactions, nonequilibrium population dynamics, and secondary succession) and abiotic factors (e.g., climate-induced disturbances, erosion and accretion, fire, temporal edaphic variability). Anthropogenic

disturbances functioning as physical stressors, however, can alter the frequency, specificity, and intensity of change.

The most apparent effect of human-caused stressors on individual habitats is acceleration of the rate of change. For example, although extinction occurs naturally, anthropogenic disturbance greatly accelerates the rate; erosion is a natural process, but agricultural practices (e.g., clear-cutting of forests) can accelerate the process by several orders of magnitude; climate change is a natural process that may be accelerated and biased by the release of greenhouse gases.

Frequently anthropogenic physical stressors reduce habitat heterogeneity by accelerating successional processes without triggering concomitant increases in the origination of early successional habitats. Such truncation of normal processes often results in habitat change that is unidirectional and irreversible. Consider, for example, the threat to Louisiana forest wetlands posed by increased rates of subsidence, which appear to be associated with petroleum pumping and the construction of levees. The stressed habitats have become truncated and the lowland forests made vulnerable to flooding and salt-water encroachment. Areas that would have been suitable for colonization as existing forest habitats became flooded have been developed for residential and agricultural use and are protected by the same levees that may be affecting subsidence rates.

Generally the ecological effect of a physical stressor on a habitat is loss of a rare, desirable, or economically important system or species. Although the stressor may mimic natural disturbances and successional processes, additional physical stressors may combine to alter the trajectory of habitat change. For instance, in the Louisiana bottomland area the combination of lowland subsidence and the development of land at slightly higher elevations is likely to result in the loss of cypress-tupelo forests in their present area and limit the trees' ability to exploit new areas.

Thus a conceptual model should emphasize processes that influence the viability of populations and species, the maintenance of certain ecosystem functions, and the diversity of ecological communities. Also, the model should address any stress that threatens the loss of a keystone species (e.g., natural predators of deer, which are inclined to overgraze) or a keystone process (e.g., fire suppression) as well as any stress altering the natural pattern of disturbances. When ecological effects are anticipated at the level of a target species, habitat suitability indices should be used; however, when the integrity of an entire community may be affected, indices of biotic integrity should be used (Karr, 1981; Karr et al., 1986; Poulson, 1992).

3.3.2.3. Habitat Destruction and Landscape Effects

Physical stressors that promote habitat destruction differ significantly from those that merely cause habitat change. Habitat change involves a level of intrusion that is reversible to some degree because the basic functioning of the ecosystem is maintained. In contrast, habitat destruction involves change that is essentially permanent (see chapter 7, on ecological recovery). When assessing the potential for habitat destruction, a determination must be made about whether the loss of the habitat will itself act as a stressor on the broader ecosystem. Such a stress can pose unacceptable risks to regional, local, or adjacent areas. Moreover, organisms that must migrate through the destroyed habitat may be affected.

In such cases, ecological risk assessment may be in conflict with human risk assessment. In part, human risk assessment is concerned with whether natural ecosystems pose a threat to humans in adjacent areas (e.g., from disease vectors). Ecological risk assessment, however, models the potential stresses imposed by human development on the adjacent natural ecosystems. The opposite perspectives of ecological and human risk assessments with regard to habitat destruction relate to edge effects; that is, boundaries between habitats (human or otherwise) provide windows through which processes and activities within a habitat can strongly influence processes within another habitat. Indeed, the boundary itself may function as a distinct habitat promoting or inhibiting a variety of functions of adjacent communities. Through such boundary areas can pass a variety of chemical, physical, and biotic stressors. For example, habitats adjacent to farmland experience fertilizer and pesticide runoff; habitats near urban development may be subjected to chemical stressors from sewer systems and biotic stressors from domestic pets and inadvertently introduced species.

Along with edge effects, habitat destruction can promote fragmentation and insularity within ecosystems (e.g., creation of a road may divide a species community). Fragmentation and insularity can be modeled on species-area relationships, which demonstrate that as fragments of a community become smaller and more isolated they will support less diversity (e.g., chaparral isolates in the vicinity of San Diego, California, harbor far fewer bird species than areas of the same approximate size within continuous chaparral habitats [Bolger et al., 1991]). Also, small isolated populations are more susceptible to demographic and environmental stochasticity (Shaffer, 1981; Goodman, 1987) and less likely to receive immigrants or disperse successful emigrants. Further, as the amount of suitable habitat declines, a species' occupation of the remaining suitable habitat will also decline (Lande, 1987). Moreover, when habitat destruction pushes the proportion of remaining suitable habitat below a minimum area, extinctions will occur (Quinn and Hastings, 1987). Theories and models of

biogeography (i.e., species-area relationships, nested subsets, assembly rules, and habitat selection) can provide the basis for conceptual models dealing with habitat fragmentation and insularity.

Habitat fragmentation and insularity also may decrease the scale, or "grain," of an environment. Coarse-grained habitats occur on scales that are large relative to the activity scale of a given individual, whereas fine-grained habitats occur on a small scale relative to the activities of an individual (Levins, 1968; Brown and Pavlovic, 1992). Fine-grained habitats tend to promote generalist species to the exclusion of species that are habitat specialists. Thus habitat fragmentation and insularity may result in the loss of species that are specialized for a particular coarse-grained habitat. For example, of three woodpecker species in forests of the midwestern United States, fragmentation has resulted first in the loss of pileated woodpeckers (the largest), followed by the decline of hairy woodpeckers (intermediate size), leaving apparently unaffected only downy woodpeckers (the smallest), which reliably reside within suburbs. In Illinois, gray squirrels and fox squirrels are associated with large-scale and fragmented tracts of woodlands, respectively. A steady, statewide decline in gray squirrels and an increase in fox squirrels has been linked to deforestation, which took place up until the 1950s. More recently, however, increased forest growth has been reversing these trends. Such changes in species population trends can provide useful information for developing a conceptual model of habitat change or destruction.

Source-sink habitats represent another important ecological concept pertaining to habitat destruction (Pulliam, 1988). In many ecological systems, a species may occupy a particularly favorable habitat that supports surplus individuals available to spread from the hot spot (Goodman, 1987) into less favorable, sink habitats. Source-sink habitats may exhibit three important characteristics:

- In the absence of the source, the population would become extinct in the sink habitat;
- Under certain conditions, the sink habitat may harbor a larger population of the species than the source habitat; and
- The quality of the sink habitat influences population viability in the source habitat (i.e., a mild sink may increase viability, while a severe sink may reduce viability in the source habitat).

Thus destruction of source habitats may have serious consequences for a target species. Also, habitat destruction that transforms mild sinks into severe sink habitats may have equally negative consequences. For instance, the success of reintroducing wolves into Yellowstone National Park may hinge critically on the availability of somewhat hospitable habitat outside of the park's boundaries.

With respect to habitat destruction, any conceptual model must consider the diversity, scale, fragmentation, and linkages of habitats. The several modeling approaches available for such studies include metapopulation analyses (DeAngelis and Waterhouse, 1987; Murphy et al., 1990), source-sink population processes (Pulliam, 1988), population viability analysis (Dennis et al., 1991), species-area relationships, and nested species subsets (Patterson and Atmar, 1986). Rolstad (1991) provides a list of the conceptual issues relevant to habitat destruction and fragmentation (table 3-2).

3.3.3. *Biological Stressors*

The introduction of native, exotic, or genetically engineered organisms can constitute biological stressors on an ecosystem (see chapter 6). Such introductions may be intentional (e.g., fish stocks), casual (e.g., "Meadow in a Can" seedings introduced to urban gardens, exotic bird releases in Hawaii), or unintentional (e.g., sea lampreys and zebra mussels in the Great Lakes).

In regard to biological stressors, conceptual models should focus primarily on the adverse ecological effects of a species introduction (e.g., What are the ecological effects of the sea lampreys already present in areas of the Great Lakes?). A conceptual model concerning an introduced species should include parameters for, or at least consideration of, the organism's proclivity for dispersal, establishment, and proliferation (Barnthouse et al., 1987). The risk manager may be interested in either introducing a chemical to control or eradicate the biotic stressor (e.g., the application of lampricides) or releasing a genetically engineered organism for control (e.g., *Rhizobia* containing additional genes to enhance nitrogen fixation). For the conceptual model in such a situation, species eradication or control programs would be the management goals, the organism would be considered the source, and the organism's activities as a competitor, predator, or pathogen would represent the stressors.

Alternatively, the conceptual model might focus on the risk of invasion or spread of an organism that might become a stressor. In this case, the risk manager may be evaluating the advantage of establishing a quarantine to reduce the chances of another invasion in the Great Lakes by the zebra mussel. In the conceptual model, regulations pertaining to the transport and quarantine of plants and animals would be the management goals, activities such as shipping would constitute the sources, and activities such as pumping bilge water or transporting organisms would represent the stressors that may have the ecological effect of introducing an undesirable exotic species.

Table 3-2. Conceptual Issues and Modeling Considerations under Habitat Alteration, Fragmentation, and Destruction (adapted from Rolstad, 1991)

Habitat Change	Consequences
Population-level considerations	
Reduced connectivity, insularization, increased intrafragment distance	Reduced or less effective dispersal, reduced immigration, less vagueness about space, increased environmental stochasticity, reduced population sizes, disruption of source-sink process
Reduced fragment size, reduced total area	Increased demographic stochasticity, increased extinction rates
Landscape or community-level effects	
Reduced interior-edge ratio, reduced fragment size or frequency	Favor generalist over specialist species; increased competition, predation, and parasitism from species in surrounding areas; increased extinction rates
Reduced heterogeneity within fragments	Loss of species diversity, increased dominance by a few species, reduced population sizes for most species
Loss of keystone species from the habitat	Loss of species or change in community composition via indirect effects such as apparent competition, resource competition, and indirect mutualism

Note: These are typical habitat disturbances resulting from human land-use patterns. Each of these sorts of disturbances (habitat change) can be modeled using theories from population ecology that alternatively (or in combination) include spatial and temporal variability, habitat selection, life-history parameters, density-dependence, environmental and demographic stochasticity, and population interactions such as competition and predation.

4. ECOSYSTEM COMPONENTS AT RISK

Summarizing available data on ecosystems at risk constitutes the second step of problem formulation. In evaluating such data, the risk assessor must:

- Identify the ecosystem and its components that are at risk;
- Select the levels of biological organization at which to anticipate ecological effects; and
- Identify the possible ecological effects.

Ecosystems involve interactions between organisms and the biotic and abiotic components of their environment—interactions that can be altered by stressors such as exposure to chemicals, habitat disturbances, harvesting, and species introductions. Thus the risk assessor must consider cause-and-effect linkages between stressors and components of the ecosystem. Because such linkages require the co-occurrence of the stressor and the ecosystem, an important issue concerns the spatial and temporal distribution of the stressor (see section 3.2.1.5.). The assessor must determine whether the scale of the agent is geographically vague (e.g., as with granular carbofuran) or fixed in time and space (e.g., as with subsidence of the Mississippi delta).

Once the geographic bounds of a stressor have been determined, the habitats within the geographic range can be specified (e.g., do the stressor's characteristics involve aquatic or terrestrial systems?) with the following considerations in mind:

- The habitat specificity of the stressor determines the ecosystem components, which can be characterized as abiotic or biotic, that may be subject to primary effects. Habitat specificity may be considered in terms of adverse effects to an entire habitat or community (e.g., from clear-cutting a forest) or in terms of particular taxonomic groups within a community (e.g., chemical stressors may selectively affect plants, vertebrates, or invertebrates).
- Linkages between stressors and ecosystem components also can be defined in terms of primary effects. The primary effect concerns the point of entry of a stressor into an ecosystem and the components of the ecosystem that are directly influenced by the stressor. Primary effects may be abiotic (e.g., levee construction resulting in subsidence, a chemical stressor changing water pH) or biotic (e.g., acute effects of carbofuran on birds). Generally, primary effects will lead to a cascade of indirect or secondary effects. Whether greatest emphasis is placed on primary or secondary effects in the conceptual model depends on the magnitude of primary and secondary effects, the ecosystem components at risk from these effects, available data, and which effects are considered adverse or undesirable.

- Biotic components can be further characterized in terms of the level of biological organization, since the response of biotic components occurs at the biochemical, physiological, behavioral, population, and community levels. By selecting particular levels of biological organization as the focus of the conceptual model, the risk assessor is not suggesting that other levels of organization are not important or may not be subject to effects; in fact, effects at one level have a high likelihood of producing effects at other levels of biological organization. Rather, the risk assessor should select the level of relevance for which data do or can exist and for which there is agreement and support for the ecological principles underlying the effects. By considering several levels of organization, a conceptual model may provide independent verifications of adverse effects, since the concordance of effects across several levels of organization builds confidence in the risk assessment.

4.1. Ecosystems Bounds

The process of defining an ecosystem at risk involves establishing the discrete bounds for communities of interacting species while recognizing that some ecosystem linkages occur on a global scale. Although such definitions of ecosystem components are primarily concerned with operational characteristics, spatial patterns relating to abiotic conditions of climate and geology are also relevant. At the global scale, for example, the earth can be divided between land masses and the oceans, but the continental shelves and tidal zones are important intermediary ecosystems.

The concept of zoogeographic realms has been used to describe the tendency of plant (and animal) species to be specific to particular areas (e.g., this explains why Hawaii is home to a disproportionate number of the species listed under the Endangered Species Act). In contrast, the concepts of biomes and life zones have been used to describe distributions of plant communities at the scale of continents. Temperature and precipitation provide reliable predictors of vegetation type, structure, and productivity. Thus, for example, increasing precipitation can cause shifts in plant communities along a continuum of xerophytic scrub, grassland, savanna, and forests.

At the scale of states, assessment of vegetation types can be further improved by incorporating the effects of soil type, geology, and hydrology. The predominant materials in soils and the interactions of these materials with plant communities strongly influence important abiotic soil conditions such as acidity, alkalinity, nitrogen and phosphorus content, and soil particle size (e.g., soil properties may either exacerbate or mitigate the effects of acid rain). At the scale of counties within states, further characterization of vegetation types can be made by considering past and present

human land-use practices (e.g., agricultural, urban development). At still smaller scales, incorporation of slope and aspect from topographical maps can be used to assess plant communities.

Moreover, available data can be used to characterize vegetation communities (e.g., data from satellite imagery or geographic information systems [GIS], national parks surveys, or municipal arboretums). The level of detail in classifying possible vegetation communities at risk will be influenced by available data and the spatial scale of the stress regime.

Because of its relative conspicuousness and direct association with particular abiotic factors, vegetation (i.e., primary producers) when studied can provide considerable information for characterizing an ecosystem. Other aspects of the ecosystem often are tightly correlated or associated with the primary producers (e.g., soil microbes, fungi, and invertebrates as well as vertebrate herbivores and carnivores). Some range maps can provide information on birds and mammals; many amphibians, reptiles, and fish; and a few insects and other invertebrates. Besides professional databases, natural history surveys and surveys performed by groups such as the Audubon Society can provide valuable information on species distributions.

Within a particular ecosystem, it may be possible to further focus the sites of primary effects by making distinctions between aquatic and terrestrial systems; the air, soil surface, and rhizosphere; and habitats or microhabitats. For example, chemical releases into air, soil, water, or sediments may be inferred from fate and transport models.

4.2. Level of Biological Organization

The source and characteristics of the stressor can provide indications about ecosystem components at risk and the spatial and temporal scale of potential effects. Thus the next step in developing the conceptual model involves selecting the specific components of the ecosystem that are likely to be sensitive to the stressor's primary or secondary effects. Stressors can affect an organism in terms of biochemistry (intracellular processes), physiology (homeostatic processes occurring among cells and organ systems within multicellular organisms), behavior (feeding, mating, and vigilance responses of an individual), population (density, age/stage-structure, and growth rate among individuals within a species), community (assemblages or food webs of interacting populations of different species), ecosystem (abiotic and biotic processes that determine nutrient cycling and energy flow), or landscape (processes occurring among tenuously coupled populations or communities).

In general, the selection of ecosystem components integrates three considerations:

- Ecological significance (see chapter 2) and regulatory aspects;

- Amount and types of existing information; and
- Potential for ecological effects based on current scientific theory.

The selection of particular levels of organization directs the conceptual model toward particular fields of study in ecology and environmental toxicology. Biochemical and physiological effects will draw heavily from toxicology and pathology; data will come from laboratory measurements of effects and examination of individuals from the site for indications of lesions, organ malformations, or tissue breakdown. Behavioral effects draw heavily from the fields of ethology and behavioral ecology in which observed or inferred behaviors of organisms at risk can be compared to norms developed from known natural history or expectations of theory. Population through landscape effects draws from the appropriate subdisciplines of ecology. Population effects specifically can be modeled and assessed through life-history models in which a stressor may influence age- or stage-specific mortality or reproductive schedules. Community effects can be modeled through theories of population interactions in which primary effects on one species may influence its predators, prey, and competitors, and where effects on one or several species may introduce changes in the dynamic properties of the community or alter successional trajectories of community development. Landscape effects consider how a stressor or its impact may become transferred among habitats or communities of an ecosystem. Such transfers occur because communities are often coupled through the migration of individuals or the transfer of chemical stressors via water, air, or soil media. Landscape ecology provides relevant theories for modeling these effects (e.g., metapopulation dynamics [DeAngelis and Waterhouse, 1987; Murphy et al., 1990]).

4.3. Ecological Effects

A wide range of possible ecological effects can result from any given stressor. At one extreme are ecosystems that are precariously balanced and highly susceptible to even slight perturbations. At the other are ecosystems that are highly resilient and predetermined. For many ecosystems, however, only certain components are susceptible to change in response to a given disturbance, while other components are quite resilient.

Although most assessment endpoints will concern ecological effects that have consequences at the level of populations or communities, in most cases ecological effects at or below the level of individuals should have less relevance in the conceptual model except as measurement endpoints. Of particular significance are ecological effects that directly influence mortality or fecundity within a community because such effects can have consequences at the population scale. Effects that increase

mortality include those that interfere with an individual's developmental or homeostatic processes (sometimes referred to as chronic effects) and those that remove individuals from the population (i.e., acute effects). Life-history models of population growth can be used to determine which changes in age- or stage-specific birth and death rates will have significance for population growth. When appropriate, consideration of density dependence can help the risk assessor evaluate whether changes in population growth rates will be mitigated by compensatory mortality or fecundity. Population growth dynamics can be particularly relevant when the stressor is a chemical substance that may have direct inhibitory or stimulatory effects on the ecosystem or the stressor is a physical disturbance involving exploitation and harvesting. In the study of granular carbofuran, it was found that consumption of a single granule could be fatal to a small bird, since 5-day subacute LC_{50}s indicated high toxicity among bird species despite variability. In the Louisiana forest wetlands example, the FORFLO model of forest dynamics provided a formal life-history presentation of mortality and fecundity effects.

Another view of ecological effects recognizes the energy and resource needs of organisms. For example, dams can restrict the access of salmon to spawning grounds. Similarly, subsidence in the Louisiana forest wetlands restricts the access of adult trees' root systems to oxygen and deprives seeds and seedlings of suitable germination and growth sites. Thus change in access to resources can be expected to have population consequences. Resource-based models of population dynamics are useful for evaluating the effects of factors such as light, temperature, carbon dioxide, and moisture on plant life; the effects of forage quantity and quality on herbivores; and the effects of predators on fishery stocks. The resource-based approach also may be appropriate for modeling the effects of chemical stressors that act as fertilizers or physical disturbances that result in the removal of resources (e.g., how might arctic whales be influenced by the harvesting of krill?) or restrict access to resources (e.g., how is the cheetah's hunting success influenced by the presence of tourists?). In the Ohio contaminated waste site example, the coke plant's lagoons harbored benthic macroinvertebrate faunas indicative of moderate to extreme eutrophication.

The risk assessor also must recognize that effects on one component of an ecosystem can have cascading effects on other components. For example, the connection between food webs can result in indirect effects. Thus a predator of a prey that has been exposed to inhibitory chemicals may suffer in two ways: (1) the predator may become exposed to the chemical via prey consumption (e.g., biomagnification), and (2) the availability of prey may decline if the prey begins to die off because of the chemical exposure.

More generally, several important indirect effects are associated with population interactions:

- Resource competition can occur when two predators share the same prey;
- Apparent competition can occur when two prey share a common predator;
- Indirect mutualism can occur when two predators have different prey that are themselves competitors; and
- Trophic cascades can occur when an organism preys on the predator of another prey species.

These effects can have relevance for modeling.

Resource competition models the negative effect of one species on another via a shared prey or resource (Tilman, 1982). Increasing the abundance of one species depresses the availability of the resource, which in turn reduces the growth rate and population size of the other competitor species. Evidence suggests that harvesting whales has resulted in more Antarctic penguins as a result of greater krill availability to penguins via reduced whale populations. In the conceptual model, any stressor that increases or decreases the population size of a species can be expected to influence the population size of its competitors.

Apparent competition models the negative effects of one species on another via a shared predator (Holt, 1977; Holt and Kotler, 1987). Increasing the abundance of one prey species increases the number of predators, which decreases the survivorship and population size of the other prey species. Marine bivalves and gastropods (despite the fact that they occupy entirely different feeding niches) have been shown to have a negative effect on each other because each attracts lobsters and octopus predators (Schmitt, 1987). In the conceptual model, any stressor with a primary effect on one species can be expected to have secondary effects on other species that share the same predator.

Indirect mutualism models the positive effect of two predator species on each other. The positive indirect effect occurs when the predators have different prey species that are competitors. Increasing the abundance of one predator reduces the population of its prey, which increases the population of the other predator's prey. For example, two aquatic herbivores, snails and daphnia, are apparent mutualists. Snails feed on periphyton (attached algae), while daphnia filter-feed for phytoplankton. The snails, by reducing periphyton, free up nitrogen and phosphorus for the phytoplankton, which feeds the daphnia. For this reason, laboratory populations of snails and *Ceriodaphnia* do better when grown together than when grown apart (Rovelstad, personal communication).

Trophic cascades model the positive effect that a predator may have on its prey's resource (see Kerfoot and Sih, 1987, and papers therein). A predator increases the abundance of its prey's resource by reducing the prey's population size or by making the prey less efficient as foragers. For

example, the threat of predation from lions may prevent zebras from overgrazing the vegetation in their range. Trophic cascades can lead to the paradox of enrichment (Rosenzweig, 1971) in which improving the resources available to a prey species may result in more predators rather than more prey. As a result of trophic cascades, adverse effects of a stressor on a resource may become manifested at the level of the resource-consumer's predator and not at the level of the resource consumer. Considerations of trophic cascades in a food web conceptual model can help direct attention to those trophic levels that are most likely to be influenced by a stressor's indirect effects.

5. SELECTION OF ENDPOINTS

As noted in the *Framework Report*, endpoint selection is a critical component of the problem formulation stage of a risk assessment. The *Framework Report* as well as the literature on risk assessment (e.g., Suter, 1989; Suter and Barnthouse, 1993) define assessment endpoints as formal expressions of the environmental attributes protected by management actions and addressed in risk assessments. In contrast, measurement endpoints are the specific laboratory and field observations that provide the empirical basis for the risk assessment.

Defining assessment and measurement endpoints for particular ecosystems involves three steps:

- Identifying the valued attributes of the environment that are considered to be at risk;
- Defining these attributes in operational terms; and
- Selecting the types of data required to assess the status and potential changes in those attributes (Suter and Barnthouse, 1993).

As noted in chapter 2 on ecological significance, the identification of "valued attributes" is a complex process involving sociopolitical considerations. The principal concern of the discussion in this section is the translation of values into endpoints rather than definition of the values themselves.

Valued attributes are derived from regulations, which in some cases define the process for identifying the attributes. For example, the NEPA "scoping process" is used to identify public and agency concerns for evaluation in environmental impact statements. In contrast, the Clean Water Act (section 316b) stipulates that a "balanced indigenous population" must be maintained in water bodies receiving thermal plumes from power plants.

In most cases, agencies have relatively wide latitude for translating statutorily defined values into assessment endpoints. Even under the Endangered Species Act, the U.S. Fish and Wildlife Service has substantial flexibility in defining the meaning of the term "species" and in prioritizing the lengthy list of potential candidate species (see chapter 2).

Because there are many possible assessment endpoints for most ecological risk assessments, the framework specifies three factors to be considered in making a selection: ecological relevance, policy goals and societal values, and susceptibility to the stressor.

5.1. Ecological Relevance

Ecologically relevant endpoints "reflect important characteristics of the system and are functionally related to other endpoints" (U.S. EPA, 1992, page 13). Since determining ecological relevance in specific cases requires a great deal of expert judgment, clearer definitions of "important

characteristics" and "functionally related" are needed if this particular criterion is to have any practical value. Suter and Barnthouse (1993) have cast some light on the definition of relevance by elaborating on the concept of significance:

> The biological significance of a property is determined by its importance to a higher level of the biological hierarchy. For example, a physiological change is biologically significant if it affects a property of the whole organism such as survival or fecundity, a change in fecundity of individuals is biologically significant if it affects the size, productivity, or other property of the population, and a decrease in the size of a population is biologically significant if it affects the number of species, the productivity, or some other property of the ecosystem. [page 23]

Certain major categories of organisms (e.g., principal primary producers, forage species, keystone predators) and ecosystem processes (e.g., primary production, nutrient cycling) can be generally defined as biologically significant according to the above definition. For generic risk assessments, however, no more specificity in the definition may be required. Yet for (certain) risk assessments, determinations of which population or ecosystem characteristics are ecologically significant must be made on a site-specific basis from expert judgment and preliminary site surveys.

5.2. Policy Goals and Societal Values

Because risk assessments are performed to support management decisions and managers must be sensitive to societal concerns, organisms that are considered endangered or of commercial or recreational value are obvious choices as assessment endpoints regardless of their biological significance. Indeed, because they are rare, endangered species would not necessarily be considered biologically significant according to the Suter and Barnthouse (1993) definition. Thus, although the definition of relevance is important in ecological risk assessment, it is equally important for ecological concepts to be communicated to managers and the public at large. That is, assessments of risks to species diversity or ecosystem function can influence decisions only if the value of these characteristics can be effectively communicated. It is essential that guidance be provided on how these values can be communicated through conceptual models—guidance that should be established at the agency level as well as the program level.

5.3. Susceptibility to the Stressor

Determining susceptibility to the stressor involves considering both real and potential exposure to a stressor as well as the types of effects that can result. For chemical stressors, properties can be used to predict the environmental partitioning and degradation effects and the ecosystem components that will be exposed. Also, toxicity tests can be used to predict which of the potentially exposed organisms are likely to be most sensitive. For other kinds of stressors, susceptibility may be more difficult to determine. Based on general life-history considerations, it is known that populations with long life cycles and low reproductive rates are more vulnerable to extinction from increases in mortality than are species with short life cycles and high reproductive rates (Mertz, 1971; Barnthouse, 1993). Susceptibilities of species to extinction caused by landscape fragmentation are often related to their home range and minimum habitat size. Guidance on determining susceptibility to nonchemical stressors is needed both at the program level and at the assessment level.

5.4. Scale Considerations

Additionally, specific ecosystem types and taxonomic groups influence the selection of assessment endpoints in terms of scale considerations. Because all organisms are destined to die, individual organisms are valid assessment endpoints only if they belong to species protected by statute. Otherwise, effects must occur over a large enough area to have an adverse impact on biological hierarchy or have a noticeable impact on populations or ecosystems of interest to the public. For this reason, most assessment endpoints should be defined at the population or ecosystem level rather than at the individual level. At the higher levels of organization, the time-space scales addressed are likely to be suited to observations and experiments performed to support risk assessments (Suter and Barnthouse, 1993).

5.5. Measurement and Assessment Endpoints

Measurement endpoints, as defined both in the *Framework Report* and by Suter (1993), are quantities such as LC_{50}s or diversity indices measured by toxicologists or field biologists. Some of these values may be generically defined (e.g., water-quality criteria) for use in standardized assessments; others are defined and measured on an assessment-specific basis.

Since endpoints generally refer to characteristics of populations and ecosystems defined over fairly large spatial scales and long time periods, it is usually impractical to directly measure changes in these characteristics as part of an assessment. In many cases (e.g., pesticide registration and toxic

chemicals review), assessments must be made before any large-scale release can be allowed to occur. Thus, some form of extrapolation based on expert judgment, statistical methods, or simulation models is usually required to link the measurement endpoints to assessment endpoints. Because assessment endpoints will usually be keyed to levels of biological organization above the individual organism and because measurement endpoints frequently are set at the level of individuals, effects on individuals usually need to be aggregated or extrapolated to infer effects on population or ecosystem-level assessment endpoints.

5.5.1. *Endpoints for the Granular Carbofuran Study*

While guidance for endpoint selection can make the process seem fairly straightforward, in practice the process can be complex. Consider, for example, the case study for granular carbofuran (Houseknecht, 1993). FIFRA requires simply that pesticides pose no "unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs, and benefits of the use of the pesticide" (FIFRA, section 2[bb]). Specific definitions of "unreasonable risk" were left for EPA to develop when implementing regulations. Risks to birds through direct mortality, indirect mortality (i.e., secondary poisoning resulting from consumption of contaminated prey), and reduced reproduction are identified as factors that should be considered in the pesticide registration process. Guidance is not provided, however, on whether the assessment should address risks to individual birds or bird populations, or whether risks to some nonendangered species (e.g., starlings, introduced agricultural pest) are more acceptable than risks to others (e.g., the wood thrush, a migratory species native to North America that is experiencing population declines over most of its range).

No rationale for the approach to endpoint selection is provided in the granular carbofuran case study. The assessment departed significantly from the routine because it was a "special review"—a retrospective assessment conducted after observations of bird mortality associated with applications of granular carbofuran. Several different types of measurement endpoints were employed in the carbofuran special review: observations of bird mortality after routine applications of granular carbofuran, measurements of carbofuran residues or acetylcholinesterase inhibition in carcasses of dead raptors, experimentally determined oral LD_{50}s, and field studies. Extrapolation from measurements to the assessment endpoint—"risk to bird populations"—was accomplished through simple decision rules and professional judgment. Observations of carbofuran residues or acetylcholinesterase inhibition in dead birds were assumed to indicate carbofuran poisoning. LD_{50} estimates (mg/kg) were compared to application rates (mg/ft^2) under the assumption that the presence

of carbofuran in a concentration of more than a few LD_{50}s per square foot indicated a significant risk to ground-foraging birds.

No attempt was made to draw inferences concerning potential reductions in abundance or an increase in the risk of extinction in local bird populations. Also, no case was made that mortality to individual birds is inherently unacceptable. The real, although unstated, assessment endpoints for this study appear to have been (1) the obvious mortality of birds in and around fields following application of the pesticide and (2) the secondary poisoning of raptors, especially bald eagles. The measurement endpoints for the first assessment endpoint were (1) the bird kill incidents reported to EPA, which provided an association between granular carbofuran and bird kills, and (2) the confirmatory toxicity tests and field tests, which showed that birds foraging over even very small areas (i.e., 1 square foot or less) could easily ingest lethal doses of the pesticide. Qualitative findings of granular carbofuran residues and cholinesterase inhibition in raptor corpses accounted for the only measurement endpoints used to address the second assessment endpoint.

5.5.2. *Endpoints for the Louisiana Forest Wetlands Study*

A different approach to endpoint definition was taken in the Louisiana forest wetlands case study (Brody et al., 1993). In this case, the assessment and measurement endpoints appear to have been influenced strongly by the availability of two quantitative assessment tools: the FORFLO model, which is a forest succession model (Pearlstine et al., 1985; Shugart, 1984) that simulates the influence of a hydrologic regime on the growth and reproduction of wetland forest tree species (i.e., the model considers competition among the trees and the tolerances of the various tree species to waterlogging), and the U.S. Fish and Wildlife Service HSIs, which relate the physical and biological characteristics of an ecosystem to the habitat requirements of specific wildlife species.

The case study paper identifies "physical alteration or change in the forest community and associated habitat value" as the assessment endpoint. Yet only forest trees and vertebrate species for which habitat requirements are well defined were included in the assessment. Vegetation types in the study area were characterized as "dry bottomland," "wet bottomland," and "swamp"—each with its own characteristic assemblage of tree species. The FORFLO model was used to simulate successional changes and transitions in these assemblages produced by different rates of subsidence (leading to increased flood duration and frequency). "Habitat value" was operationally defined as the habitat requirements of five representative vertebrate species: gray squirrel (*Sciurus carolinensis*), swamp

rabbit (*Sylvilagus aquaticus*), mink (*Mustela vison*), downy woodpecker (*Picoides pubescens*), and wood duck (*Aix sponsa*).

Measurement endpoints were defined simply to be the input data required by FORFLO and the HSIs. For FORFLO, these data included (1) baseline field data on the community composition of representative forest stands, (2) species-specific growth and physiological tolerance parameters, and (3) a time series of environmental data (in particular, data on hydrologic regimes). For the HSIs, the required data included various measures of species-specific habitat variables (e.g., vegetation composition, mast production, availability of herbaceous cover, standing water area).

The assessment itself proceeded in three steps: (1) scenarios for future subsidence rates were developed; (2) future changes in vegetation states were calculated based on the scenarios for three representative tree communities (dry bottomland hardwood, wet bottomland hardwood, and swamp forest) found in the area; and (3) changes in community structure (as predicted by FORFLO) were used to calculate changes in habitat suitability for the five representative species.

5.5.3. *Endpoints for the Inactive Coke Production Plant*

Endpoints for the coking plant assessment were determined principally from the ARAR that triggered implementation of the National Contingency Plan (see section 1.1.3). Because the Clean Water Act mandates protection of aquatic biota, the coking plant assessment focused on aquatic community integrity as an assessment endpoint. Three different types of measurement endpoints were employed, with Federal and State water-quality standards serving as surrogate measurement endpoints. The presence of contaminants in concentrations that exceed these criteria constitutes legally enforceable grounds for action. In addition, field surveys of benthic macroinvertebrates, fish, and amphibians were conducted to directly measure the conditions of aquatic communities at contaminated and uncontaminated sites. Finally, histopathological studies were performed on fish to determine the presence of anomalies indicating contaminant exposure.

The assessment relied on a weight-of-evidence approach to determine whether plant-derived contaminants were present in concentrations sufficient to harm aquatic communities.

6. THE CONCEPTUAL MODEL

As noted in section 1, the conceptual model serves as a summary of the risk assessment's problem formulation phase and as a plan for the analysis and risk characterization phases. We suggest including the following elements, at a minimum, in a conceptual model:

- A flow chart linking the stressor(s) being evaluated to the assessment endpoints;
- A set of discrete hypotheses concerning the possible effects of the stressor(s) on the endpoints (the hypotheses should highlight the specific measurements or model outcomes needed to distinguish among alternatives);
- The specific measurements and extrapolation techniques to be used in the analysis phase of the assessment; and
- Model/data selection criteria and quality assurance standards.

Whether this information is codified in program guidance or developed separately for each assessment depends on the specifics of the assessment being performed.

6.1. Flow Diagrams

Figures 3-1 and 3-2 present two possible forms of a flow chart linking stressor(s) to endpoints of concern. The event-tree diagram in figure 3-1 depicts various mechanisms, both anthropogenic and natural, that could contribute to recruitment failure in a fish population. This particular example of an event-tree diagram has been used by EPA's Office of Toxic Substances to illustrate its approach to ecological risk assessments (discussed in Suter, 1993). The advantage of the event tree is that it facilitates consideration of alternative mechanisms leading to the same ecological consequence. Moreover, it readily accommodates information on behavioral, physical, and toxicological effects. Although convenient for conceptualization, the event-tree diagram can be difficult to relate to suitable quantitative assessment approaches.

Figure 3-2 is a more conventional flow chart based on the physical movement of a toxic contaminant from a source, through environmental media, to direct and indirect effects on a fish population (i.e., food-chain effects). This type of flow chart has the advantage of corresponding more directly to the quantitative environmental fate models that are often used in risk assessment. It is only applicable, however, for chemical stressors. Also, it does not lead as directly to consideration of alternatives.

The flow chart shown in figure 3-3 depicts the influence of hydrology on the structure and function of a bottomland forest ecosystem (Brody et al., 1993), drawn in "energy circuit language"

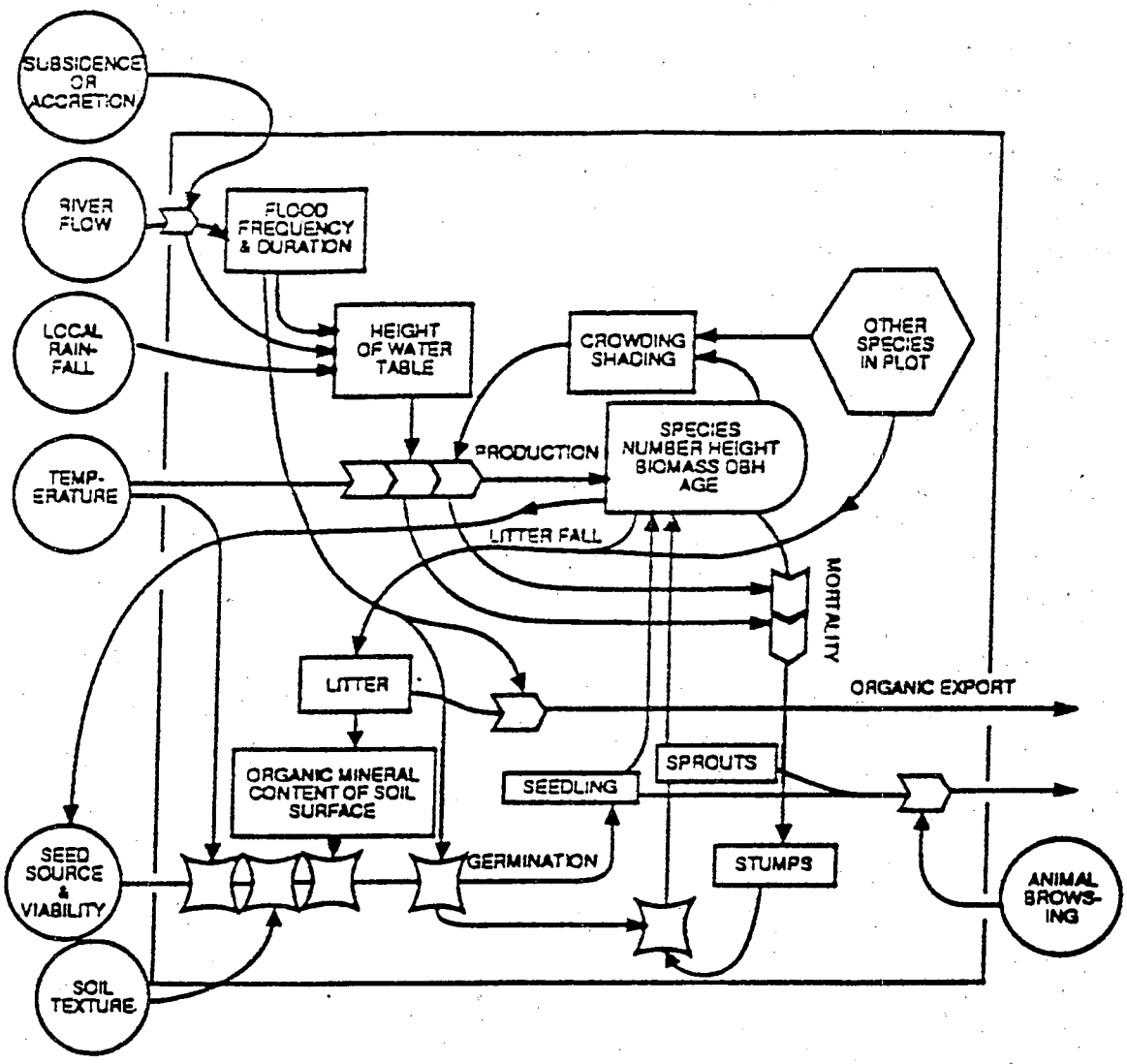

Figure 3-3. Dynamics contained in FORFLO (Pearlstine et al., 1985).

(Odum, 1971). In this chart, the various symbols represent energy sources and transformation processes, environmental influences, and internal regulatory mechanisms. Energy circuit notation is quite general, allowing representation of chemical, physical, and biological processes. A person familiar with the notation can extract from the diagram the essential features of a quantitative model of the energy sources, transformations, and sinks for the system being represented. Such diagrams are often quite complex, however, and are not readily comprehensible to nonexperts.

Regardless of the approach chosen, the problem of aggregation is the most significant technical issue to be considered in developing a flow diagram. Indeed, an infinite array of possible conceptual models could be developed for any assessment problem. For example, species could be grouped according to trophic levels or identified individually. Similarly, all possible environmental pathways could be included in the model or only those few believed to be ecologically significant based on an expert evaluation of existing information. In general, however, only pathways thought to be sufficiently important for collection of data should be included. The degree of resolution of the compartments should likewise be geared toward the data that will be collected or that are already available.

For the granular carbofuran study a highly simplified flow diagram was used (figure 3-4a). Although the diagram leaves out most of the complex ecological processes that occur in agricultural ecosystems, it provides a reasonable representation of the conceptual model actually used in EPA's special review of this pesticide. The only assessment endpoints identified are ground-foraging birds and the raptors that prey on them. The exposure pathways of concern are (1) ingestion of granules by ground-foraging birds, (2) ingestion of contaminated invertebrates, and (3) secondary poisoning of raptors feeding on poisoned prey. The only relevant environmental fate data are pesticide application rates. The only biological data employed are bird kill data (i.e., incident reports and experimental field trials), laboratory toxicity studies, and results from raptor autopsies. Figure 3-4b shows a slightly more complex flow diagram that includes ecological processes that could have been investigated but that were deemed irrelevant to the special review. This diagram, which includes a more detailed representation of the exposure process and considers raptor population dynamics, would be appropriate if the assessment had required quantitative estimates of the impacts of granular carbofuran on raptor populations.

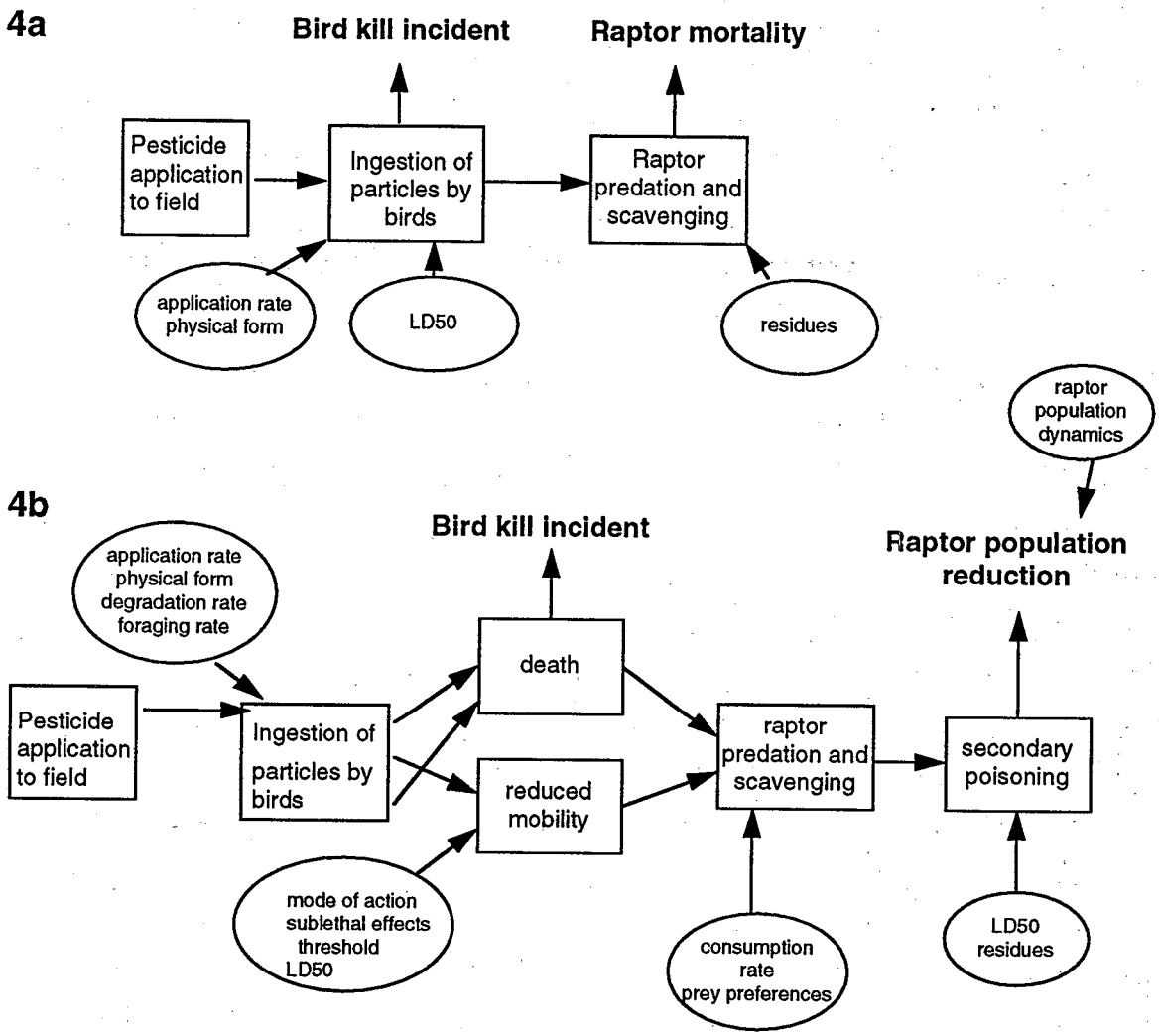

Figure 3-4. **(a) Flow diagram and conceptual model for the granular carbofuran case study; (b) expanded flow diagram for the carbofuran example.**

6.2. Impact Hypotheses and Quantitative Response Relationships

The development of impact hypotheses before conducting field studies has been recommended as a means of increasing study efficiency (Sanders et al., 1978; Beanlands and Duinker, 1984). An imaginative ecologist can list dozens—or even thousands—of potential ecological effects from any stressor. A manageable set of the most likely exposure pathways and effects must be selected for investigation during the problem formulation phase of the risk assessment. For example, figure 3-1 indicates that recruitment failure in a fish population may have many causes, some of which may be related to the regulatory action being contemplated and some of which are unrelated. The conceptual model should identify which of these causes will be investigated.

6.2.1. *Scientific Methodology for Developing the Conceptual Model*

The conceptual model should result in the development of explicit hypotheses, which will improve the efficiency of data collection and force the risk assessor to establish a plan for the assessment before getting started. Moreover, the assessor should justify the selection of particular exposure pathways and ecological effects, explaining why they are deemed of greater concern than other possible pathways and effects. Also, the conceptual model should include specific criteria for determining the significance of results (i.e., the rejection or acceptance of the null hypothesis [H_0]). By being explicit about the various components of the conceptual model, the risk assessor facilitates communication among interested parties.

To develop the model, the risk assessor should use available information on stressors, ecosystem components, and ecological effects to generate hypotheses. Even when only incomplete data are available, it is important to develop theories about connections between possible stressors and ecological effects. Nonetheless, the amount of information should strongly influence one's confidence in conclusions drawn from the conceptual model. For example, conclusions based on sparse information should be reconsidered if countervailing evidence becomes available. As available information increases, the assessor's confidence in the new or modified model should also increase.

When sufficient information is available, however, hypotheses should be explicit. In the granular carbofuran study, for example, one of the endpoints of concern was mass kills of ground-foraging birds that ingest pesticide particles. Alternative hypotheses could be stated as follows:

H_0: Granular carbofuran is hazardous to ground-foraging birds.

H_1: Granular carbofuran is not hazardous to ground-foraging birds.

The hypotheses are stated in the reverse of the usual form (i.e., a null hypothesis of no impact) because FIFRA regulations specify that a pesticide must be assumed hazardous unless risk assessments show that it is safe. H_0 would be rejected and H_1 accepted if (1) few or no incidents of bird kills associated with granular carbofuran application were reported to EPA, (2) laboratory studies demonstrated that granular carbofuran has low toxicity, or (3) field tests indicated that exposure and mortality are low under actual field conditions. Subsequently, numerous bird kill incidents were reported to EPA, laboratory studies showed that very small doses of granular carbofuran (i.e., as small as a single granule) are sufficient to kill small birds, and birds were killed in significant numbers during field tests. The null hypothesis was accepted, and granular carbofuran was declared hazardous to birds.

Davis and Bascietto (1993) noted that the planners of the coke production plant study did not state a specific hypothesis. In part for this reason, the study failed to address some potentially important assessment endpoints and exposure pathways (e.g., migratory birds, piscivorous wildlife) and failed to include some potentially useful measurements (e.g., contaminant body burdens, sediment toxicity tests). A hypothesis for the endpoint that *was* investigated can be stated as follows:

H_0: Contaminated water and sediment are not adversely affecting aquatic biota.

H_1: Contaminated water and sediment are adversely affecting aquatic biota.

In this case, the no-impact hypothesis is the null hypothesis because CERCLA mandates action only if contaminants are released in quantities large enough to be hazardous to human health or the environment. Because this study employed a weight-of-evidence approach, rejection criteria are not self-evident. Additionally, because water-quality standards are legally enforceable, H_0 would presumably be rejected and H_1 accepted if those standards were exceeded. If no standards were exceeded *and* the bioindicators (e.g., diversity indices, fish tumors) measured at "exposed" stations were not significantly different from those measured at "reference" stations, then H_0 would be accepted. An ambiguous result would occur if bioindicators showed differences between exposed and reference stations but no water criteria were exceeded. Rules for "weighing the evidence" in weight-of-evidence assessments should be stated (and agreed on by all interested parties!) before performing the assessment. Otherwise, the results will be open to conflicting interpretations.

Although in principle it should be possible to state every risk assessment in the form of explicit hypotheses and tests, in practice this approach is not always practical for decision making. Consider, for example, the Louisiana forest wetlands case study. Rather than a regulatory assessment of the acceptability or unacceptability of a specific product or proposed action, this assessment involved

quantitative estimation of potential ecosystem effects associated with likely changes in the rates of subsidence. Further development of this study would presumably identify quantitative relationships between specific watershed management strategies and future subsidence rates. Agencies responsible for watershed management would use this information to make more informed decisions about future land and water use in the lower Mississippi floodplain. Assessments in which managers are presented with a quantitative relationship rather than a pass/fail criterion are common in resource management and in NEPA assessments. Rather than specifying null and alternative hypotheses, conceptual models for these kinds of assessments should specify (1) the specific relationships to be included (e.g., between subsidence and tree community structure) and (2) the data required to develop those relationships.

6.3. Summary of Measurement Endpoints and Extrapolation Techniques

The measurement endpoints and extrapolation methods for testing impact hypotheses or developing relationships should be listed, referenced by hypothesis, and identified with the appropriate compartments and links in the flow diagram. While these steps may appear trivial, they are important for reviewers of assessment plans because they encourage the risk assessor to be explicit (i.e., avoiding such rationales for data collection as "it's always done this way").

In some regulatory programs, specific measurements (e.g., toxicity test protocols) or extrapolation equations are established in program guidelines. In other situations, these are developed on a case-by-case basis. In Superfund investigations, for example, compilation of a list of data requirements is the initial stage in the establishment of data quality objectives (DQOs); this list is (or should be) referenced to a flow diagram. Once the types of data are identified, specific measurement techniques can be selected and statistical confidence limits established. The development of procedures for establishing DQOs (an ongoing effort within the Superfund program) provides a means of bridging the gap between conceptual model development as described in the framework and the specific sampling plans required for field investigations.

6.4. Model/Data Selection Criteria and Quality Assurance Standards

As noted above, specific models and data types can be specified in program guidance only for standardized assessments. For programs in which most or all risk assessments are site specific, criteria for selecting models and ensuring data quality should be established in program guidance. For commonly used environmental fate models and standard environmental data types, such as

contamination concentrations and population abundance surveys, these criteria can be readily specified. For more complex studies of problems, such as introduced organisms or habitat alteration, which will require novel models and field studies, existing standards for model selection and data quality are unlikely to be adequate. When these standards exist, however, criteria for determining the adequacy of models and sampling schemes proposed for individual assessments should be included in program guidance.

Data quality considerations are especially important for assessments like the coke production plant study that are based on weight-of-evidence evaluation of field data. Issues involved in data quality determination for field studies include (1) adequacy of the proposed measurement methods, (2) comparability of "exposed" and "reference" sites, and (3) statistical aspects of sampling design. All of these issues are more easily addressed for environmental contaminant measurements than for ecological measurements.

For contaminants, "adequacy" means choosing an analytical technique that measures the quantity of interest to assessors (e.g., dissolved concentration vs. bulk concentration; total metal concentrations vs. specific oxidation states), that has been shown to produce reliable results in the medium of interest, and that is sensitive enough to detect contaminant concentrations of interest for the assessment (e.g., concentrations that would exceed regulatory criteria). "Comparability of sites" for chemical analyses means that the results of the analyses can be validly used to distinguish between sites that have been contaminated and sites that have not been contaminated. Background conditions, such as sediment oxidation state, that affect the results of chemical analyses might, for example, differ between sites believed to be contaminated and sites proposed as references. "Statistical aspects of sampling design" for chemicals means the selection of a sampling scheme that can adequately characterize spatiotemporal patterns in contaminant distribution.

"Adequacy" relates to the capability ("power" in statistical terminology) of a design to detect a particular degree of difference between groups of observations ("control" and "affected," "before" and "after") or to determine whether the concentration of a contaminant exceeds a given value. Depending on the particular application, the spatial distribution of contaminants, the temporal distribution, or both may be of primary interest. Each of these objectives leads to a different optimal sampling scheme.

All three of the above issues should be addressed by expert environmental chemists and statisticians during the development of sampling plans involving environmental contaminant measurements. These three issues are equally relevant to biological measurements, but they are often

much more difficult to resolve. For example, there is no general consensus concerning which biological measurements are "adequate" for a given circumstance. Community indices such as the Index of Biotic Integrity (Karr et al., 1986; Karr, 1992) are now commonly used as measures of aquatic ecosystem status, but substantial disagreements still exist among competent professional ecologists as to the applicability of a given index to particular systems or even as to the value of biotic indices in general (Suter, 1992). Similarly, a consensus is lacking concerning the circumstances in which in situ toxicity testing is appropriate and about the interpretation of molecular and histopathological indicators. In the absence of regulatory guidance, a variety of biological measurements are being used in site-specific ecological risk assessments. Experience from these assessments may well permit the development of consensus and guidance within a few years.

Selection of ecologically comparable "exposed" and "reference" sites continues to be a major difficulty for sampling programs designed to detect change caused by a contaminant source or any other spatially localized stressor. Nature is neither a laboratory nor an agricultural experiment station; treatments cannot be randomly assigned to replicate ecosystems. Despite criticisms of the application of classical experimental design concepts to ecological assessment (Hurlburt, 1984; Loehle and Smith, 1990), obvious alternatives do not exist. Innovative techniques such as intervention analysis (Carpenter et al., 1989) and Bayesian inference (Reckhow, 1990) avoid assumptions about replication but have not been sufficiently tested in assessment applications to determine whether they truly provide more useful results. Assuming that *any* field study will involve an attempt to compare sites exposed to different stress levels, preliminary field surveys of possible reference sites should be conducted to ensure that these sites are ecologically as similar as possible to the putatively exposed sites. Criteria for evaluating site similarity can and should be developed and standardized.

The most intractable statistical design issues for biological measurements are mentioned in the preceding paragraph. In other respects, statistical design for biological measurements is analogous to design for contaminant measurements. The principal difference is that statistical power for biological measurements is often much lower, because of higher variability and lower sample size and (depending on the measurement) lack of independence of the measurements themselves. Preliminary estimates of variability can be used to predict the magnitude of change likely to be statistically detectable in a full-scale field study.

6.5. Example Conceptual Model: Granular Carbofuran

As discussed by Houseknecht (1993), granular carbofuran is "a broad-spectrum insecticide and nematicide registered for control of pests on 27 agricultural crops and for certain forest and pineseed orchard uses." The assessment endpoints addressed in the risk assessment included lethality to ground-foraging birds and raptors preying or scavenging on ground-foraging birds.

Flow diagram: A flow diagram is depicted in figure 3-4a. Granular carbofuran is applied to the soil to kill pests that attack plant roots. Ground-foraging birds ingest pesticide particles, receiving a dose that is dependent on the application rate, the quantity of active ingredient in each particle, and the rate of degradation of the pesticide in typical soil conditions. Depending on the size of the bird and the number of particles ingested, birds may die following ingestion of one or several particles. Because carbofuran is a neurotoxin, death, if it occurs, happens shortly after exposure; if a bird does not die, it is expected to recover. Dead or behaviorally impaired birds may be eaten by raptors, which can be poisoned by eating the tainted prey.

Impact hypotheses: The null hypothesis for this assessment, defined by statute, is that granular carbofuran is hazardous. The hypothesis can be rejected if bird kills are not reported, or if laboratory studies show that carbofuran has low toxicity (i.e., kills are from some other cause), or if field tests show that exposure and mortality should be rare under actual field conditions.

Measurement endpoints and extrapolation techniques: Measurement endpoints for this assessment include (1) reports of bird kill incidents associated with carbofuran applications, (2) laboratory toxicity tests using several bird species, and (3) experimental applications of granular carbofuran under conditions meant to duplicate actual agricultural applications. A hazard index ($LD_{50}s/ft^2$) was used as a descriptor of ecological risks associated with carbofuran applications.

Model/data selection criteria and quality assurance standards: Selection criteria and quality assurance standards were not described in the case study report.

6.6. Example Conceptual Model: Louisiana Forest Wetlands

In this assessment (Brody et al., 1993), the stressor is changes in water-level elevations (0.5 cm/yr) due to subsidence, reduced sedimentation, salt-water intrusion, and reduced drainage gradients within the Lake Verret Basin. The sources of these changes in water level include natural processes (i.e., delta formation and tectonic subsidence) and anthropogenic processes (i.e., levee construction, dredging, land development, shipping, and petroleum pumping). The amount of suspended sediments in the Mississippi River may have declined by 50 percent since the 1950s. The assessment does not

purport to establish a direct link between a particular source and the stressor. Rather, it is noted that before levee construction, the Lake Verret Basin was experiencing a net accretion of sediments in which bottomland forests were actually replacing swamp forests. The shift from net accretion to net subsidence implicates the previous levee construction as the source of these changes. The assessment endpoints are (1) the area of dry bottomland forest, wet bottomland forest, and cypress-tupelo swamp forest; (2) the relative abundance, sizes, and densities of 16 canopy tree species; and (3) two species of bird (the downy woodpecker and the wood duck) and three mammal species (the gray squirrel, swamp rabbit, and mink).

Flow diagram: The FORFLO model predicts changes in forest compositions along with changes in hydrologic parameters, and it provides the flow diagram for the risk assessment (figure 3-3). Several abiotic factors (i.e., subsidence, river flow, and rainfall) influence water tables, flood frequencies, and flood durations. The water-table and flood characteristics influence the mortality and growth of the different tree species. The composition of the forest is further elaborated as a life-history model that takes the current size structure of the forest and projects seed production, seedling growth, and the future size and species composition of the forest. Continual change in flood characteristics prevents the model from achieving a stable forest composition. The flow diagrams leading to the HSIs are not given in the case study.

Impact hypotheses: The null hypothesis for this assessment is that the present forest types and compositions within the Lake Verret Basin will remain unchanged into the future. This hypothesis will be rejected if the output of the FORFLO model shows a regular succession of forest types from bottomland hardwoods to swamp forest. In one sense the null hypothesis is a "straw man" because increased water levels will inevitably lead to swamp and open-water habitat. In the absence of a dialogue between the risk manager and the assessor, however, it is impossible to structure the impact hypotheses in the form of upper and lower bounds of acceptable habitat change. More generally, the output from FORFLO and the HSIs represent a set of testable hypotheses.

Measurement endpoints and extrapolation techniques: For the abiotic components, measurement endpoints include height of the water table, flood frequency, and flood duration (from the U.S. Geological Survey). For the species composition of the forest, measurement endpoints include tree species frequency, dominance, density, size, and replacement sequences (measured over a 2-year period). Temperature and precipitation estimates were taken from climatological records. Other parameters for the tree species (e.g., maximum growth rate, optimal water level) were taken

from estimates found in the published literature. The necessary parameters for the bird and mammal HSIs were derived from FORFLO, measured directly in the field, or estimated from published values.

Model/data selection criteria and quality assurance standards: Selection criteria and quality assurance standards were not described in the case study report.

6.7. Example Conceptual Model: Inactive Coke Production Plant

Stressors for this assessment (Davis and Bascietto, 1993) consisted of organic and inorganic contaminants released by the coke production plant during its years of operation. The assessment attempted to (1) identify these contaminants in surface water and sediment potentially exposed to those releases and (2) detect changes in biotic communities potentially affected by contaminant exposures. Assessment endpoints consisted of fish community structure, fish health, and invertebrate community structure.

Flow diagram: A flow diagram is depicted in figure 3-2. Contaminants released by the plant are deposited in lagoons, stream sediments, and wetlands, where they reach chemical equilibrium between sediment and water and biological equilibrium with aquatic biota. Contaminants may affect benthic invertebrate communities and aquatic communities by favoring pollutant-tolerant species over pollutant-sensitive species; fish that survive may have impaired health. The diagram serves as a useful tool for summarizing the kinds of data that would be useful for confirming the presence or absence of exposures and effects. Note that certain pathways occur in figure 3-2 that were not addressed in the case study: wetland vegetation and fish-eating vertebrates. If a sampling and analysis plan for the coke plant study were being prepared today, regulatory agencies reviewing figure 3-2 would probably question the absence of vegetation and wildlife studies at the site.

Impact hypotheses: The null hypothesis for this study is that contamination of water and sediment by pollutants derived from the plant have had no effects on fish and invertebrate communities in potentially exposed aquatic ecosystems. The hypothesis would be rejected if water-quality standards for protection of aquatic life were exceeded for any of the measured contaminants. Fish and invertebrate community data were used to support results from water-quality analyses; however, no criteria for rejecting the null hypothesis based on these biological data were presented.

Measurement endpoints and extrapolation techniques: The study attempted to measure contaminant effects in three ways. First, contaminant concentrations measured in surface water were compared to EPA water-quality criteria. Second, fish and benthic invertebrate community structures at contaminated and uncontaminated sites were compared. Third, fish livers were examined for

tumors indicative of contaminant exposure. Standard statistical techniques were used to test for differences between "contaminated" and "uncontaminated" sites; no dose-response models or other quantitative measures of impact were employed.

Model/data selection criteria and quality assurance standards: No selection criteria or quality assurance standards were described in the case study. Reviewers of the study (Davis and Bascietto, 1993) noted numerous deficiencies in study design, including (1) failure to address important assessment endpoints such as waterfowl and mammals, (2) use of insufficiently sensitive chemical analysis methods and fish health indicators, (3) inappropriate selection of reference sites, and (4) use of inadequately documented community indices. It should be emphasized that at the time the coke plant investigation was performed, no technical guidance on ecological risk assessments at CERCLA sites was available. Current procedures for development and review of sampling and analysis plans would have identified many of the above deficiencies before field studies were initiated.

7. SUMMARY

The flow chart in figure 3-5 describes one possible process for developing a conceptual model using the concepts discussed in this chapter. Two aspects of the figure require further explanation. First, conceptual model development often needs to be an iterative process. Thus, after the initial attempt at problem classification and data summarization, a "consistency/completeness check" should be performed to determine whether significant aspects of the problem or the data have been misconstrued or left out. An additional consistency/completeness check should be performed after initial assembly of the conceptual model. Existing peer-review processes used by EPA and other regulatory agencies already provide a mechanism for this kind of review and revision. Second, many if not most risk assessments involve an initial screening step that filters out obviously negligible risks before the development of assessment plans involving expensive toxicity testing or field data collection.

The principal themes of this chapter can be summed up as completeness, efficiency, and scientific rigor. Risk assessments should be *complete* in terms of addressing all relevant management goals and considering all ecological knowledge necessary to support informed ecological risk management decisions. The discussion of management contexts in section 2 is intended both to emphasize that the definition of "complete" differs for different kinds of management problems and to identify the types of program-level risk assessment guidance appropriate for different regulatory situations. The discussions of stressor characteristics and ecosystems at risk in sections 3 and 4 are intended to summarize the range of stressor-specific and ecosystem-specific information of potential value for ecological risk assessments and to provide criteria for identifying the information required for any given assessment.

Risk assessments should be *efficient* in terms of collecting only relevant information and avoiding information that is redundant. Necessary information should not neglected. Further, the time required for the assessment should be commensurate with the time available for making risk management decisions. Section 2 discusses the influence of available time on decisions concerning the balance between modeling and site-specific data collection in risk assessments. Sections 3 and 4 discuss how knowledge of the stress regime and ecosystem components can bring focus to the broad range of potential ecological effects. The selection of endpoints (section 5), the development of flow diagrams (section 6.1), and the summary of measurement endpoints and extrapolation techniques (section 6.3) are all intended to facilitate efficiency.

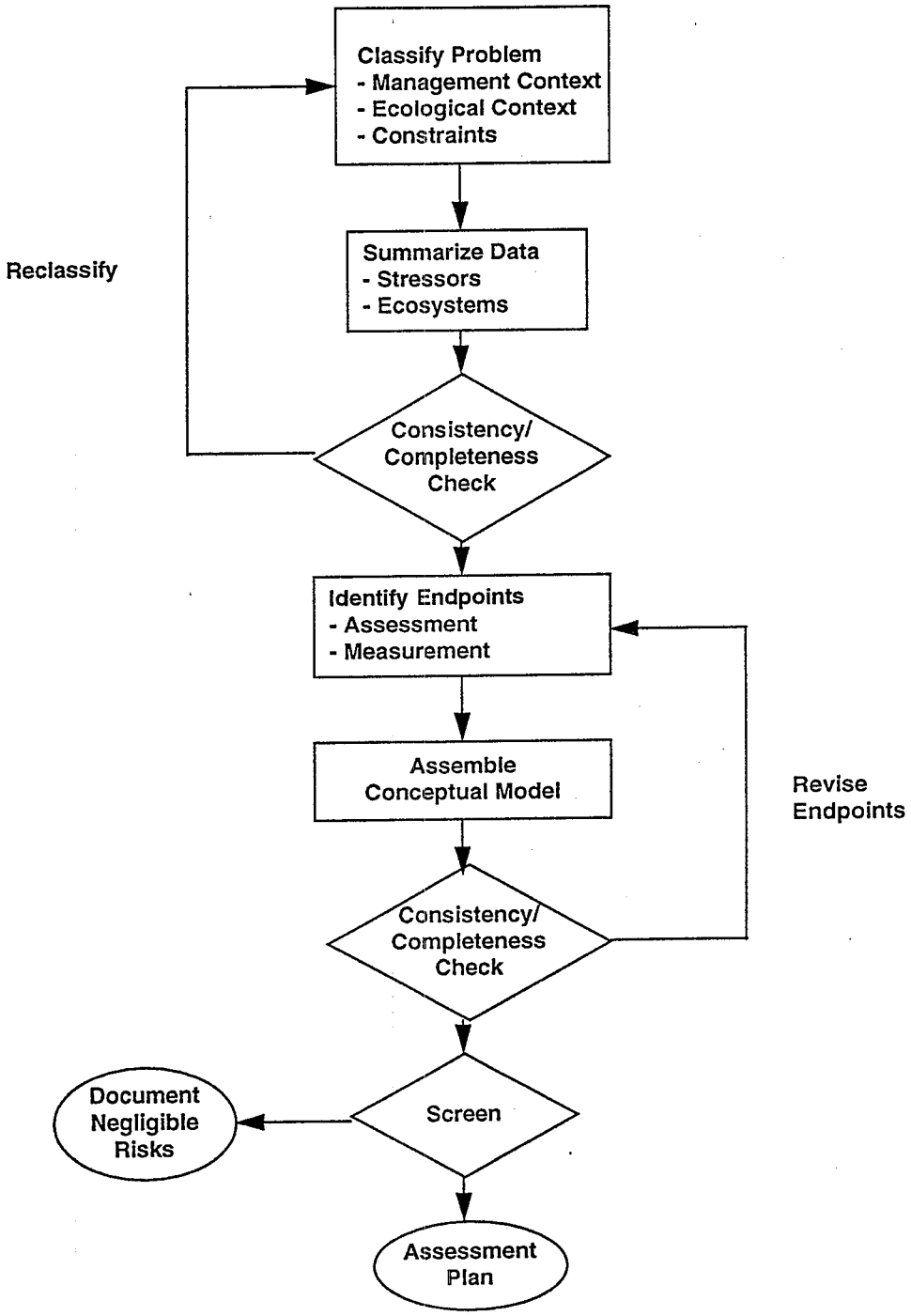

Figure 3-5. **Flow chart for development of a conceptual model. The scheme includes two checkpoints for consistency/completeness and allows for a screening-level assessment that identifies negligible risks before initiating intensive data collection or analysis.**

Risk assessments obviously should be scientifically as *rigorous* as possible, given the constraints imposed by the needs of risk managers for timely decisions. Risk assessment cannot be expected to be as rigorous as basic scientific research, however, because research is generally not constrained by timetables and research scientists can (and should) be extremely conservative in determining the amount of evidence required to accept or reject hypotheses. Although assessment scientists have no such luxury, they still should be expected to establish and meet appropriate standards of scientific rigor. The discussions of impact hypotheses (section 6.2) and data/model selection criteria (section 6.4) are directed at facilitating the establishment of such standards.

With reference to the *Framework Report* (U.S. EPA, 1992), a properly developed conceptual model should ensure completeness, efficiency, and rigor by summarizing all of the technical elements of the assessment in a form that enables risk managers to determine whether management objectives will be addressed and that permits scientific peer reviewers to determine whether standards for technical adequacy will be satisfied. While the development of a complete and credible conceptual model does not guarantee a successful risk assessment, the absence of such a model virtually ensures a risk assessment with useless or indefensible results. We have tried to identify the range of issues that must be addressed and the process required to develop good conceptual models. It is still up to each agency or program office to use these principles to develop guidelines and quality standards appropriate to its specific mission.

7.1. Future Needs

Three specific areas for guidance development were identified during the preparation of this chapter. Guidance is needed on methods for determining the relative susceptibility of different types of populations and ecosystems to nonchemical stressors such as physical habitat change. Such methods would be analogous to the qualitative structure-activity relationships and short-term toxicity tests that ecotoxicologists use to screen potentially hazardous chemicals. Screening-level risk assessments are essential for focusing available assessment resources on those situations that are most likely to pose significant risks. Screening-level risk assessment procedures are widely used in risk assessments for chemicals and should be developed for nonchemical stressors as well.

Guidance on using a weight-of-evidence approach for risk characterization is needed to eliminate an important source of ambiguity in ecological risk assessments. In principle, ecological risk assessments can be greatly strengthened by using independent types of data and models (e.g., contaminant analyses, in situ toxicity tests, biotic indices) to address the same endpoints. Often,

however, the different lines of evidence can produce conflicting results because each type of data has its own unique sources of bias and uncertainty. Guidance on interpreting these conflicting results is needed to prevent risk assessments from being selectively interpreted by parties with vested interests in potential risk management actions.

Finally, guidance on the use of conceptual models for communication with risk managers and the public is needed. The National Research Council (1993) has noted that communication between ecological risk assessors, risk managers, and the general public must be greatly improved if ecological risk assessments are to have significant influences on environmental decisions. Conceptual models that succinctly summarize key management questions and scientific information needs can serve as effective communication tools.

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Issue Paper
on

CHARACTERIZATION OF EXPOSURE

Glenn W. Suter II
Environmental Sciences Division
Oak Ridge National Laboratory
Oak Ridge, Tennessee

James W. Gillett
Department of Natural Resources
Cornell University
Ithaca, New York

Sue Norton
Office of Health and Environmental Assessment
U.S. Environmental Protection Agency
Washington, D.C.

Prepared for:

Risk Assessment Forum
U.S. Environmental Protection Agency

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1. INTRODUCTION

The *Framework for Ecological Risk Assessment* (U.S. EPA, 1992a) and all other risk assessment paradigms require characterization of exposure. The critical concept in this phase of risk assessment is that an organism or other system must be in contact with or co-occur with an agent or a disturbance caused by an agent (i.e., it must be exposed) before there is a risk (U.S. EPA, 1992a,b).

In this chapter, we examine the process termed exposure, which extends from the release of an agent from a source to its uptake or interaction with the ecosystem or its components that constitutes the assessment endpoint.[1] Sources produce or release agents as physical, chemical, or biological entities. By analogy, the exposure concept can be applied to various human activities that physically disturb the environment. Ultimately, the purpose of this chapter is to discuss the issues relating to the development of an exposure profile as an input to the risk characterization for both chemical and physical agents. Although biological agents have a number of characteristics that conceptually parallel those of physical and chemical agents, assessing them presents particular difficulties. Thus biological agents are discussed elsewhere (see chapter 6 on biological stressors).

This introductory section defines the bounds of exposure characterization. Section 2 discusses the role of exposure in problem formulation. Sections 3 and 4 consider the characterization of exposure in traditional exposure assessments and in less traditional assessments that involve predictive or retrospective analysis. Section 5 discusses the product of the characterization of exposure—the exposure profile. Section 6 identifies some information gaps. (Terms used in this chapter are defined in the glossary, which also addresses ambiguities and controversies concerning terminology.)

1.1. Sources and Agents

This chapter draws extensively from experience with chemical substances as pollutants and various physical agents (e.g., heat, ionizing radiation) as health and ecological hazards. The addition of such agents to the environment is the usual subject of risk assessments. Except for introductions of biological agents, exposure assessments for additions of chemical agents to the environment are conceptually straightforward. The characterization of exposure is intended to apply to these as well as to many types of disturbances generated by human activities. The parallel structure of risk

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assessments for additions of agents and for actions of agents causing physical disturbances is depicted in figure 4-1.

The application of this conventional exposure terminology to physical agents that cause disturbance is less familiar. As with releases of contaminants, the impetus for the assessment is a plan, proposal, or equivalent activity that is regulated or otherwise subject to a decision involving ecological risk considerations. Examples include management plans for a fishery or forest or project plans to construct a dam or dredge a stream. These are equivalent to the plans to market a chemical for a particular use. If the plan is for a single action at a particular site, its implementation is the source. If the plan involves separate and distinct actions, however, the source addressed in the assessment is likely to be the individual actions that implement the approved plan or proposal. Examples include a timber sale that implements a forest management plan or construction of a housing development to implement a land use plan. This is equivalent to the source of a chemical such as a pesticide application, which is the implementation of a use registration, or a wastewater outfall, which is an implementation of the registration of detergent chemicals.

The agent disturbs the environment as a result of the action of a source. Examples include, in the case of the fishery plan, trawlers; in the case of stream dredging, the dredge; and in the case of a dam construction project, the dam itself. As with chemicals, the characteristics of the agent may be crucial to the effect. For example, the type of gill nets, seines, or trawls used to harvest a fishery will determine the species, numbers, and size distribution of the by-catch, and the choice of epilimnetic versus hypolimnetic discharge from a dam will determine the water quality downstream and the entrainment of fish in the turbines.

The exposure is the process by which the agent acts on (i.e., disturbs) the receptor. Examples include the removal of a benthic community by a dredge, the flooding of a riparian community by a dam, and the clearing of a woodland by bulldozers. This is equivalent to contact of a chemical by a receptor. The risk of effect is then a function of the nature and magnitude of this exposure and the exposure-response relationship for that agent and receptor. For example, the risk of reducing the abundance of a fishery by 50 percent is a function of the catch (the exposure) and the population response to loss of members of harvestable age classes (the exposure-response relationship).

The concept of disturbance depicted in figure 4-1 applies to the actions of an agent that directly affects the receptor. In addition, disturbance is commonly used to indicate the modified state of the environment to which receptors are exposed. For example, logging may disturb trees by cutting, crushing, or burning (the direct disturbance), and the resulting state of the forest constitutes a

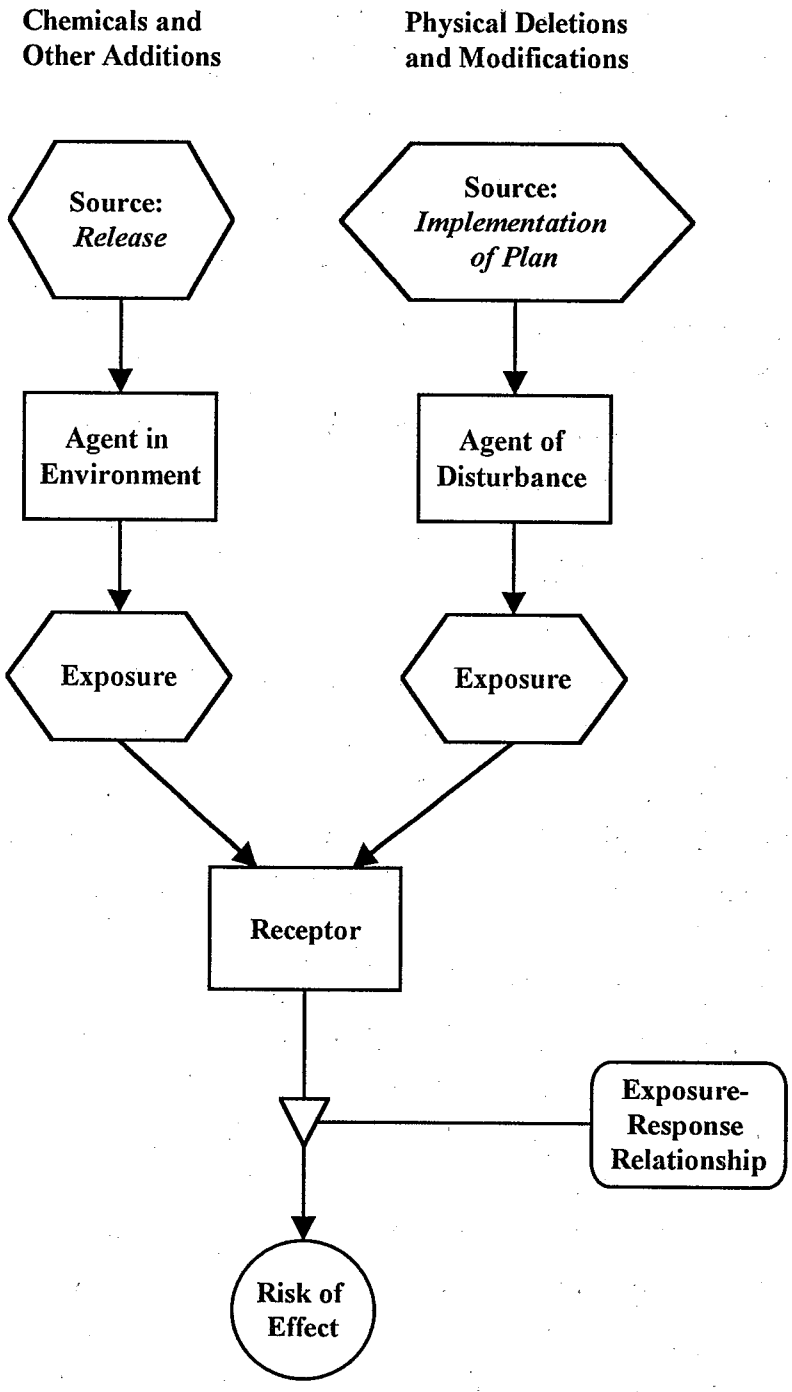

Figure 4-1. **The relationship of exposure to sources of agents, activities causing disturbances, and effects.**

disturbance to which wildlife are exposed. Because organisms do not necessarily contact deleted or modified components of the environment, we extend the concept of exposure to include co-occurrence. Such chains of primary and secondary exposures and effects occur with both chemical releases and disturbances and are discussed below (section 1.2).

Physical disturbances can be divided into three classes: system eliminations, system modifications, and component deletions. System eliminations replace an existing ecosystem with an anthropogenic system, such as a parking lot or soybean field. For these, characterization of exposure can be a simple matter of describing the area replaced and its former suitability for the endpoint species, community, or other attribute. System modifications do not completely destroy an ecosystem but modify its character (e.g., changes in the hydrologic regime of a stream by diversion or hydroelectric generation). In general, characterization of exposure for such an agent involves defining an area and its change in suitability. Component deletions remove a particular component of the ecosystem without causing an extensive physical change (e.g., resource harvesting, entrainment in power plant cooling systems). For such disturbances, the expression of exposure is the number of organisms or other units removed per unit area. For system eliminations, time is usually irrelevant because the exposure is essentially infinite; for system modifications and component deletions, however, duration and frequency must be specified.

This chapter is deliberately unbalanced in the sense that it is largely devoted to exposure to chemicals. Introductions of biological agents are treated in a separate paper, and exposures to physical agents are usually treated as specialized assessments (i.e., heat, ionizing radiation, increased ultraviolet [UV] radiation from stratospheric ozone depletion). The different types of physical disturbances are both significant and common, but they have not been treated as subjects of ecological risk assessment. Although assessments of physical disturbances have been conducted, particularly in environmental impact assessments, these assessments have been ad hoc and have not developed a consistent approach for characterizing exposure to those disturbances. Thus, in contrast to chemicals, relatively little guidance is available on characterization of physical disturbances of ecosystems.

1.2. Types of Exposures

The ultimate objective of exposure characterization is to demonstrate a logical connection between the subject source of an agent and the receptors in such a manner as to permit evaluation of the relationship. Figure 4-2 shows a typical, simple exposure process that extends from a pesticide application to exposure of birds. Although the exposure-response relationship and ultimate effect (the

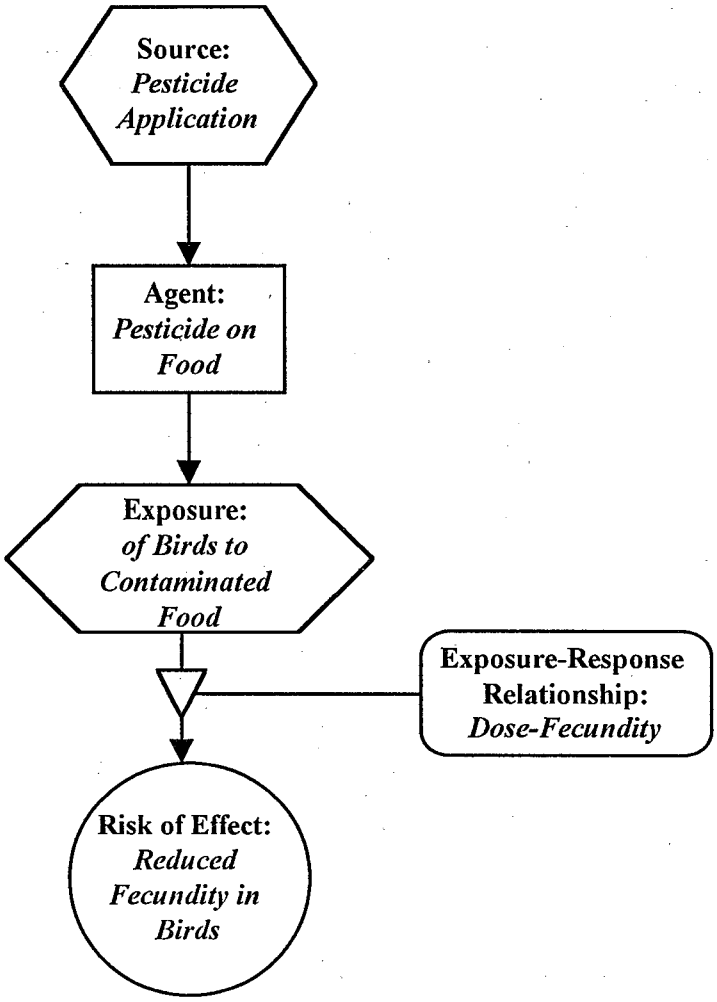

Figure 4-2. **An example of a simple ecological risk assessment involving exposure of birds to a pesticide application (the risk-of-effect box represents the assessment endpoint).**

assessment endpoint is the reduced fecundity in birds) are included in this diagram for completeness, they would be analyzed as part of the ecological effects assessment and risk integration steps, respectively.

Such a simple structure can quickly become complex when the primary agent is transformed to secondary chemical or physical agents through physical or biological processes. Examples include the increasing UV radiation brought on by the release of chlorofluorocarbons (CFCs) (figure 4-3), the lowering of dissolved oxygen (DO) after increased nutrient loads (figure 4-4), and the generation of methyl mercury from inorganic mercury by anaerobic microbial processes.

Secondary disturbances attributable to physical deletions and modifications of the environment can be handled similarly. Examples include the increase in stream temperature from the removal of riparian vegetation or the change in frequency from the flooding of riparian communities downstream of a levee (figure 4-5).

These secondary exposures logically fall within the purview of exposure characterization, and they require the expertise of exposure assessors. For example, in the case of nutrient loads to an estuary, estimation of DO levels probably would be performed by the same chemist or engineer who models dispersion of the nutrients. Indirect effects, however, have traditionally been treated in the ecological effects characterization component of an ecosystem effects model (O'Neill et al., 1982; Suter and Bartell, 1993) or more commonly are deferred for the risk characterization component (e.g., the chemical is toxic to crustacean zooplankton at estimated exposure levels; since zooplankton are eaten by fish, fish are at risk).

Combined exposure and effects models have been proposed (Bartell et al., 1988) and used in specific situations. For example, Brody et al. (1989) modeled the combined changes in plant communities and associated wildlife populations resulting from an increase in flooding frequencies following the construction of river levees. These combined models, however, are not widely used and are required only in assessments where there is significant feedback between exposure and effects. For example, if the biota are a major sink for the contaminant and toxic effects are sufficient to significantly decrease standing biomass, available concentrations will increase due to increased sorption, uptake, or biodegradation. Similarly, toxic effects may modify behavior, thereby changing exposure.

Although it is not necessary to separate the exposure characterization from the effects characterization, it is critically important that the interface between the two is clearly defined in the assessment to ensure completeness. Consider, for example, assessments of chronic exposure of

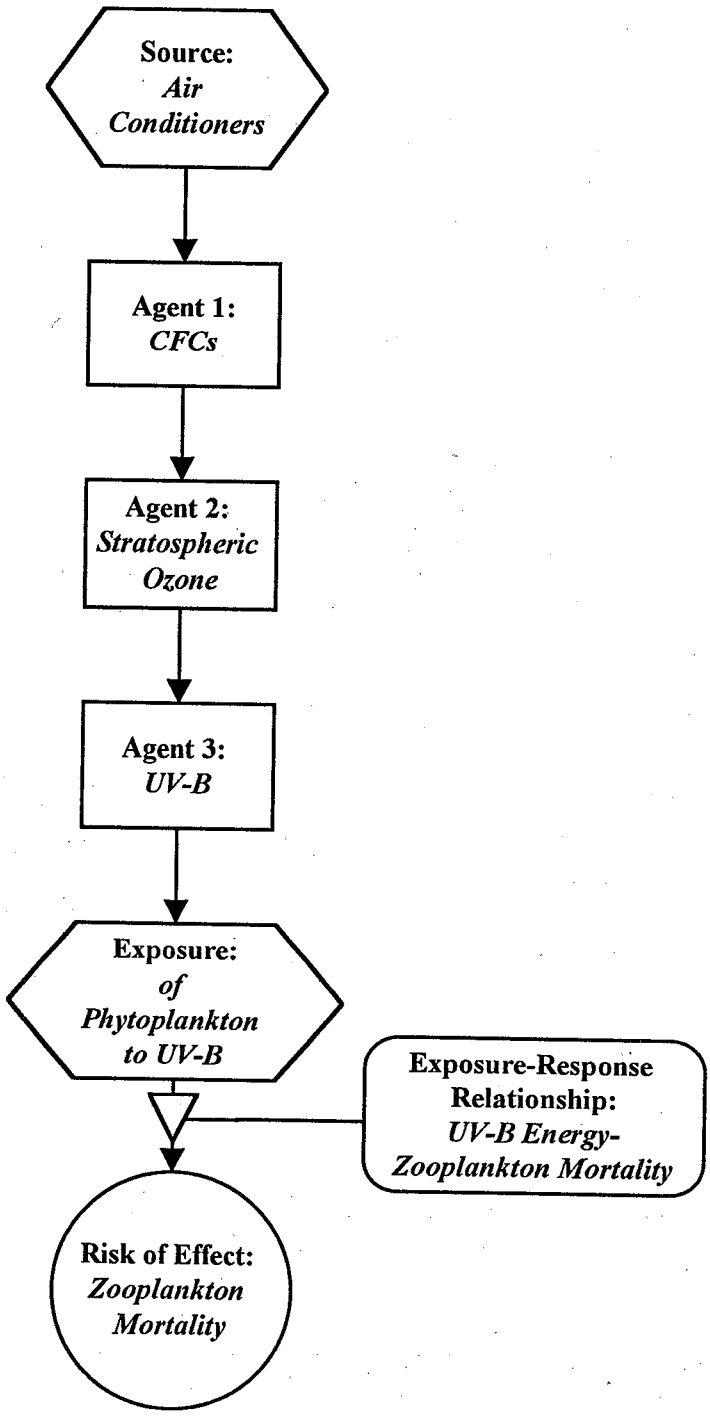

Figure 4-3. **An example of an ecological risk assessment involving a multistage generation of agents but a single exposure resulting in a direct ecological effect. The multistage agents would be an ultimate agent (UV radiation) generated by a multistage exposure process from an initial agent (chlorofluorocarbons) released by a source (air conditioners).**

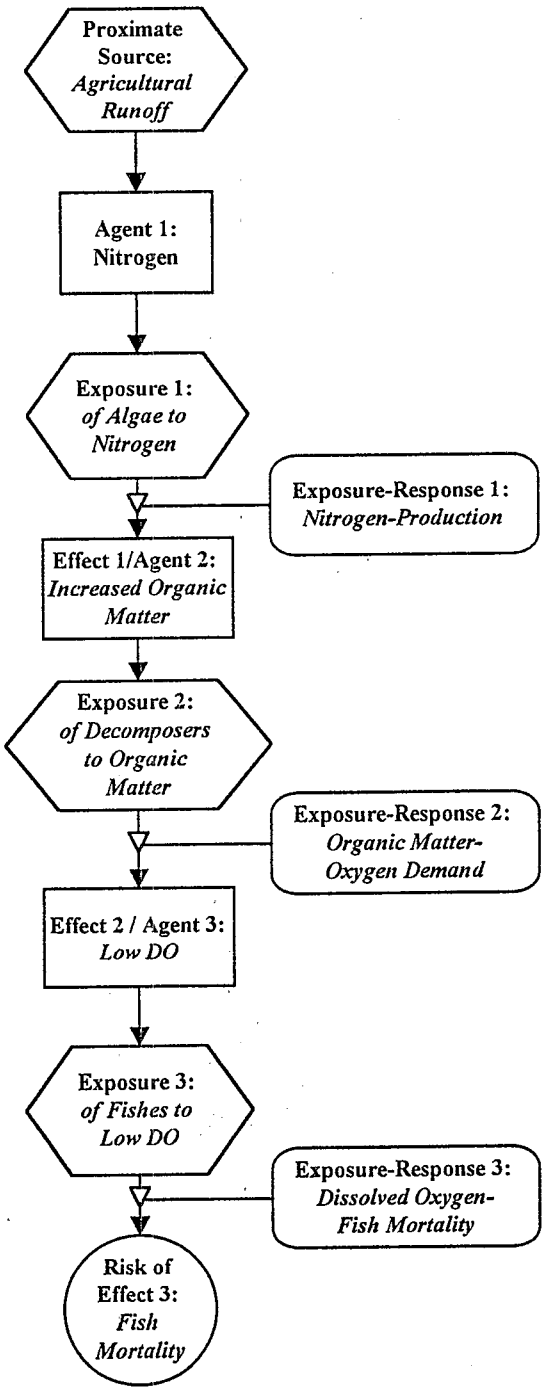

Figure 4-4. **An example of an ecological risk assessment involving a causal chain of multiple exposures and effects. This example, however, could be treated as a multistage exposure (like the risks from CFCs in figure 4-3) because the intermediate ecological exposure-response relationships are commonly incorporated into the exposure model (the risk-of-effect box represents the assessment endpoint) so that only the third exposure-response relationship comes from the effects assessment.**

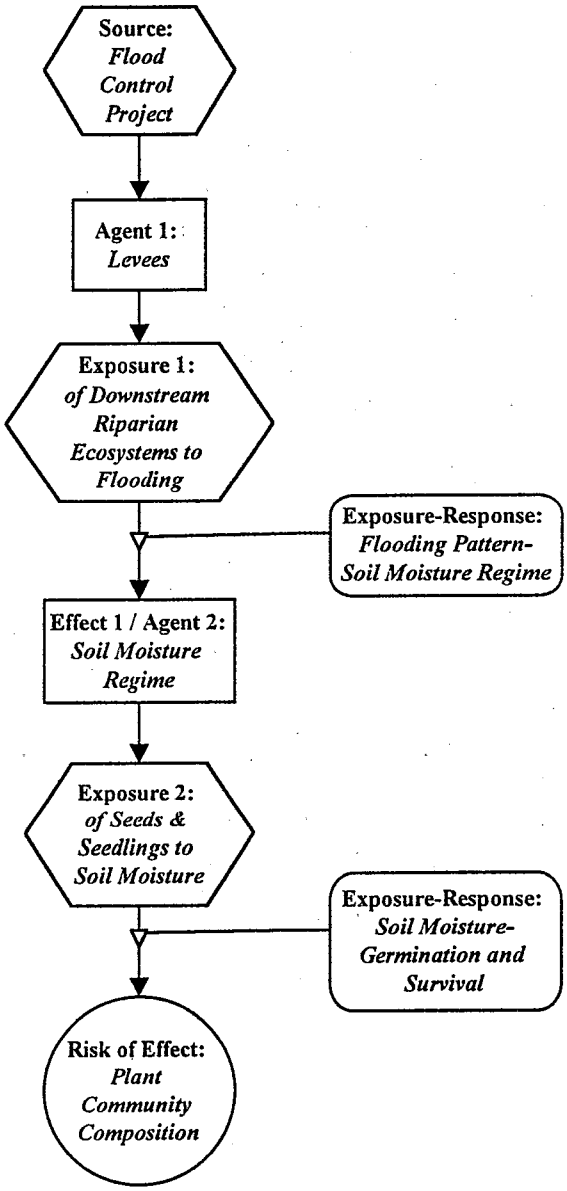

Figure 4-5. Example of an ecological risk assessment involving exposure to a disturbance generated by a multistage exposure process involving an intermediate effect on the physical habitat.

aquatic organisms to chemicals. While exposure assessors typically have seen their task as estimating the equilibrium total concentration of chemical in water, effects assessors have seen their task as estimating the continuous concentration causing effects in clean laboratory water. The issues of bioavailability and temporal dynamics often "fall through the cracks" between the two analyses. In this regard, the emphasis on an interaction between exposure and effects analyses in the *Framework Report* represents an advance over previous risk assessment paradigms.

1.3. Predictive and Retrospective Assessments

In this chapter we recognize two types of assessments. The first is predictive assessments, which estimate exposures and effects of proposed future contaminant releases or other hazardous actions. Predictive assessments begin with an estimate of the source and proceed to model the processes that result in exposure or disturbance.

The second class of assessments is retrospective assessments. Since observations or measurements of exposure and effects may be available for such assessments, epidemiological techniques can be employed (Suter, 1993). Indeed, a greater diversity of methods is appropriate for retrospective assessments than for predictive assessments. Retrospective assessments may address observed effects such as reproductive failure in certain birds and winter kill of certain salmonids; such assessments are said to be effects driven. Alternatively, retrospective assessments may, like predictive assessments, be concerned with estimating the effects of a source (i.e., source-driven assessments). An example is the assessment of previously dumped wastes that can now be remediated.

2. PROBLEM FORMULATION

Problem formulation, the initial step in ecological risk assessment, defines the threat to be assessed and thereby determines the scope and content of the assessment (U.S. EPA, 1992a). The exposure assessor must collaborate with the effects assessor to define the problem in such a way that both the potentially significant ecological threats and the concerns of the risk managers are incorporated.

The principal product of problem formulation is a conceptual model that describes the sources or actions, the agents or disturbances, the exposure pathways, the causal pathways for secondary effects, and the assessment endpoints.

The exposure analyst is primarily responsible for definition/characterization of the following:

- *Agent and source.* For predictive assessments, the agent and source are provided as an input. Nonetheless, the exposure assessor may need to characterize the source more fully for purposes of the exposure assessment (e.g., by asking for information on temporal variability of the source or by converting expected sales of a detergent builder such as phosphate into rate and distribution of releases). In retrospective assessments, the source and even the agent may be unknown and the exposure assessor may need to help define the potential sources and agents based on potential exposure pathways.
- *Assessment endpoint.* Of the criteria for selection of assessment endpoints (U.S. EPA, 1992a; Suter, 1989), the exposure assessor is most concerned with estimating susceptibility. Thus, although inherent sensitivity must be considered, the assessor should be particularly concerned with populations or communities that are the most exposed and therefore likely to be susceptible (see glossary).
- *Ecosystem at risk.* The exposure assessor has a major role in this component of conceptual model development that includes defining the locations of the sources, the extent of distribution of the agent, and the media contaminated or the communities exposed or disturbed. The exposure assessor also must ensure that the ecosystems at risk and the abiotic and biotic factors that influence the exposure are defined sufficiently for the analysis phase to be performed.

2.1. Defining the Source and Agent

The first link in the causal chain portrayed in the conceptual model is the introduction of a chemical, physical, or biological agent by a source, or an equivalent action that initiates a disturbance

(e.g., timber sales that initiate logging). The processes for defining the source and the agent are different if the assessment is initiated by concern about a prescribed source or by concern about observed effects.

2.1.1. *Source-Driven Versus Effects-Driven Assessments*

Source-driven assessments are conducted to determine the risks posed by a particular source or set of sources. Such assessments usually concern the regulation of new chemicals and effluents, establishment of criteria and standards, or remediation of spills or other past releases. Because the source serves as the impetus for the assessment, it must be identified at the outset of problem formulation as specifically and quantitatively as possible using available information. Such information might include locations, composition of the released agent, rates of release, temporal dynamics of release, mode of release, and carrier media. For source-driven retrospective assessments, such as remedial investigations of hazardous waste sites, measurements of contaminants in ambient media (including biota) may supplement source definition.

Effects-driven assessments are conducted to determine the cause of observed effects and to estimate their magnitude and extent. Exposure characterization for such assessments calls for hypothesizing causal pathways and tracing them back to sources of agents. This component of problem formulation can be based on four types of information:

- Observation of symptoms (i.e., behaviors such as ataxia, overt pathologies such as lesions, internal physiological or histological indications) and the distribution of effects among species, trophic groups, or habitats (if differential responses of organisms are observed, they may be compared to knowledge of the relative sensitivity of the taxa, sexes, or life stages in controlled exposures; if the potential agents have different modes of exposure, the habitats or trophic groups that are most affected may provide clues to the likely cause);
- Evidence of exposure from the affected organisms, including body burdens of contaminants and biomarkers of exposure, such as induction of metabolic enzymes;
- Knowledge of the various sources and disturbances that are potentially responsible; and
- Data on the spatial and temporal relationship between the effects and the potential causal sources.

Application of these approaches to determining potential causes of fish kills is discussed by Meyer and Barklay (1990).

2.1.2. *Multiple Agents*

Multiple agents can result in a common exposure or disturbance (e.g., increased flooding from upstream devegetation, channelization), a common effect (e.g., fish population reduction attributable to toxicity, habitat loss, or overfishing), or multiple effects. In general, assessment of risks posed by multiple chemical and physical agents requires a separate exposure analysis for each agent as well as an analysis of the nature and degree of interaction (see chapter 5 on effects characterization).

2.2. Defining the Spatial and Temporal Extent

Problem formulation also calls for specifying the spatial and temporal extent of the assessment to facilitate appropriate selection or development of exposure models. This facilitates proper distribution of sampling and sufficient consideration of all significant sources and effects (see chapter 3 on conceptual models). Ultimately, the exposure assessor must ensure that the spatial and temporal scale of the assessment is sufficient to include all significant sources and exposure or disturbance processes. Moreover, the assessor must ensure that the characterization of exposure encompasses all significant direct and indirect effects.

2.2.1. *Extent Based on Agent/Source*

The spatial and temporal extent of the assessment can be defined in terms of the distribution in space of sources, the duration of release from sources, and the properties of the agent that control its movement and persistence in the environment. The role of source distribution in space and time is obvious and is usually easy to specify or estimate. For individual chemicals, appropriate considerations are the distributions of production, use, and disposal.

Once an agent is released from a source, its inherent properties determine its ability to spread and persist. Any property of the agent that makes it less site specific, more capable of moving between media, and more persistent extends the temporal and spatial basis for the assessment. Gillett (1983) suggested that a combination of the log of the half-life for biodegradation (log $t_{1/2}$), log of Henry's Law constant (log H_c), and log of the octanol-water partition coefficient (log K_{ow} or log P) projects separation of the chemicals that are likely to be leachable or to present atmospheric exposure, to be sediment/soil bound, or to be bioaccumulable. That screening function for ecotoxicological concerns is valid and useful for neutral, nonionized, nonpolymeric organic molecules; yet it has limited applicability to organometallics and charged or ionized organic molecules. Nevertheless, the ability to determine the pervasiveness and persistence of any agent seems to be of high priority. Once

susceptible, valuable receptors are identified in the iterative processes of problem formulation, then the assessment effort can be focused on the temporal and spatial dynamics of effects in relation to the agent's properties. Neely (1985) has used this approach in a global model of chemical dispersion that offers a crude scoping of any chemical in conjunction with large safety factors.

The spatial and temporal extent of actions that destroy or modify ecosystems or their components is usually easily defined in terms of proximate disturbances (i.e., the area to be flooded by a dam, to be paved for a parking lot, or to be logged can be readily specified). Typically the duration of the initial disturbance is essentially instantaneous, and the main temporal variable is the time to recovery (see chapter 7). Still, some disturbances, such as implementation of a fishery management plan, may have a finite duration.

The formation of secondary agents and disturbances may expand both the temporal and spatial extent of the assessment. For example, assessments of DDT must take into account its transformation to DDD and DDE, which have different and sometimes more powerful effects on species distant from the source of the DDT. Similarly, assessments of logging practices must take into account erosion and sedimentation of downgradient streams.

2.2.2. *Extent Based on Effects*

Specifying the distribution of observed effects, of exposed endpoint populations and ecosystems, and of secondarily affected populations and ecosystems also helps determine the spatial and temporal extent of the assessment (see chapter 7 on conceptual models, chapter 5 on effects characterization). Although the area in which the primary exposure or disturbance occurs may be relatively small and the duration of the disturbance brief, the consequences may extend to the entire range of an endpoint population or ecosystem, and the duration may extend through recovery or induction of secondary exposures and effects.

2.2.3. *Extent Based on System Processes*

System processes provide the mechanisms for transport, transformation, and degradation of contaminants; for transport and fate of introduced organisms; and for secondary disturbances. Thus in problem formulation, the environmental processes and properties that are important to the exposure must be specified. For example, in assessments of the aquatic ecological risks of the release of an organic chemical from a point source, the important processes are dilution, sorption, and degradation. Thus the flow regime of the receiving system, its suspended and dissolved organic matter content, and

any limitations on the degradation rate must be specified and incorporated into the exposure or disturbance model. Some system processes may expand the spatial and temporal extent of the assessment beyond the range of the proximate exposure and effects (e.g., acid rain, increases in UV radiation attributable to depletion of the stratospheric ozone layer by CFC compounds).

2.3. Defining the Receiving Environment

The receiving environment may be an actual place where effects have occurred or a source is located, or it may be one or more generic reference environments that typify the types of environments likely to be exposed or disturbed. If the receiving environment is a particular place, then the primary role of the exposure assessor is to ensure that it is sufficiently defined to allow characterization of the exposure. The assessor accomplishes this by using existing information, contributing to development of the conceptual model, and ensuring that sufficient information is gathered in the analysis phase to characterize the exposure.

For source-driven assessments concerning the addition of chemical and physical agents, the conceptual model must include, at a minimum, qualitative judgments concerning the significant pathways from the source to the various media in which organisms or communities would be exposed. For source-driven assessments of actions that may delete or modify ecosystems or their components, the conceptual model must include both the proximate disturbance and the causal pathways likely to transmit the disturbance to other environmental components. For example, logging can increase the silt load, nutrient concentrations, and light levels in streams; thus streams as well as forests are disturbed by logging.

For effects-driven assessments, the conceptual model must include the potential routes of exposure for the affected organisms as well as the routes of transport or the causal pathways by which the contamination has occurred. To complete this process, the assessor must rely heavily on experience and judgment. For chemical and physical agents, however, simple screening models of dilution and partitioning can be used to estimate the pattern of contamination or the pathways from effects to sources.

2.4. Assessment Endpoints

Of the criteria for choosing assessment endpoints, only the estimation of susceptibility requires input from the exposure assessor. Susceptibility is a function of the inherent sensitivity of receptors to the agent and of the magnitude of exposure or disturbance. Given that the contaminated or

disturbed environments will have been defined, the relative exposure or disturbance of a receptor is a function of the way in which the receptor uses the environment.

For chemical contaminants, the following considerations are relevant:

- Organisms that occur in or feed on the most contaminated media are more exposed.
- Organisms that are exposed to a medium by multiple routes are more exposed (e.g., fish that respire and feed from water are more exposed to aqueous contaminants than organisms that only feed from water).
- Organisms that are less capable of avoiding contaminated media are more exposed.
- Organisms that have small ranges relative to the extent of contamination will have the greatest exposure of those individuals in the contaminated areas.
- Populations that have small ranges relative to the extent of contamination will have the greatest exposure of those populations in the contaminated areas.
- Organisms contacting contaminated media during sensitive life stages (e.g., migration, reproduction) are more susceptible.

For example, if a contaminant is spilled on soil, earthworms are in the most contaminated medium since they feed on the soil, are in dermal contact with it, and respire the air in soil pores; also they have relatively slow avoidance capabilities and the range of both an individual earthworm and an interbreeding population is likely to be encompassed by a single spill. By contrast, robins are exposed to the more contaminated medium only if the contaminant is biomagnified by worms relative to the soil. Robins have relatively little dermal contact with the soil, however, and it may be fairly easy for them to avoid contaminated areas since spills are likely to be smaller than the range of an individual robin and nearly all spills will be smaller than the range of a robin population. When determining susceptibility, such exposure-related considerations should be combined with information on the sensitivity of the particular species. Susceptibility then should be considered along with other criteria to determine which species properties would serve as the most appropriate assessment endpoints (Suter, 1989; U.S. EPA, 1992a).

Similar considerations should be taken into account on a case-by-case basis when determining the susceptibility of potential endpoint species or communities to agents other than chemicals. For example, neotropical warblers that breed in large tracts of undisturbed forest would be a logical focus for an assessment of the effects of forest fragmentation, since the particular species of bird is likely to co-occur with the disturbance during a sensitive life stage.

2.5. Causal Pathways in Conceptual Models

Typical conceptual models for ecological risk assessments of chemicals or physical agents resemble diagrams of transport and fate models. Indeed, the major conceptual problem in the problem formulation phase usually concerns determining the contaminant's pathways and the ecosystem components that will be significantly exposed. A simple example for a generic upland waste burial ground is presented in figure 4-6. In this case, a judgment has been made that atmospheric routes are negligible and that carnivores would not consume enough prey from the site to be significantly exposed. Judgments and compromises also have been made about the appropriate degree of aggregation. For example, all plants are aggregated because there is insufficient phytotoxicity data or plant uptake data to perform more taxon-specific assessments. Different chemical classes or types of waste might have similar or radically different pathways, depending on their physicochemical properties and modes of action. Finally, indirect effects of the proximate toxic effects are judged to be negligible in this case.

If indirect effects were judged to be potentially significant, then the conceptual model would need to include effects pathways as well as exposure pathways (see figure 3-4 in chapter 3 on conceptual model development). Regardless of the conceptual model's structure, developing such information into a diagram is particularly useful for communicating with risk managers and with members of the assessment team.

For physical disturbances, no general concept such as contaminant fate is appropriate as an organizing focus of the conceptual model. For example, a conceptual model for risks from logging posed to salmon reproduction (figure 4-7) includes a mixture of indirect disturbances and effects that are merely part of the causal web that follows from the initial disturbance. For effects-driven assessments, the conceptual model must be based on information about exposures and disturbances and any other environmental factors that could contribute to the observed effect (see, e.g., figure 3-1 in chapter 3 on conceptual model development).

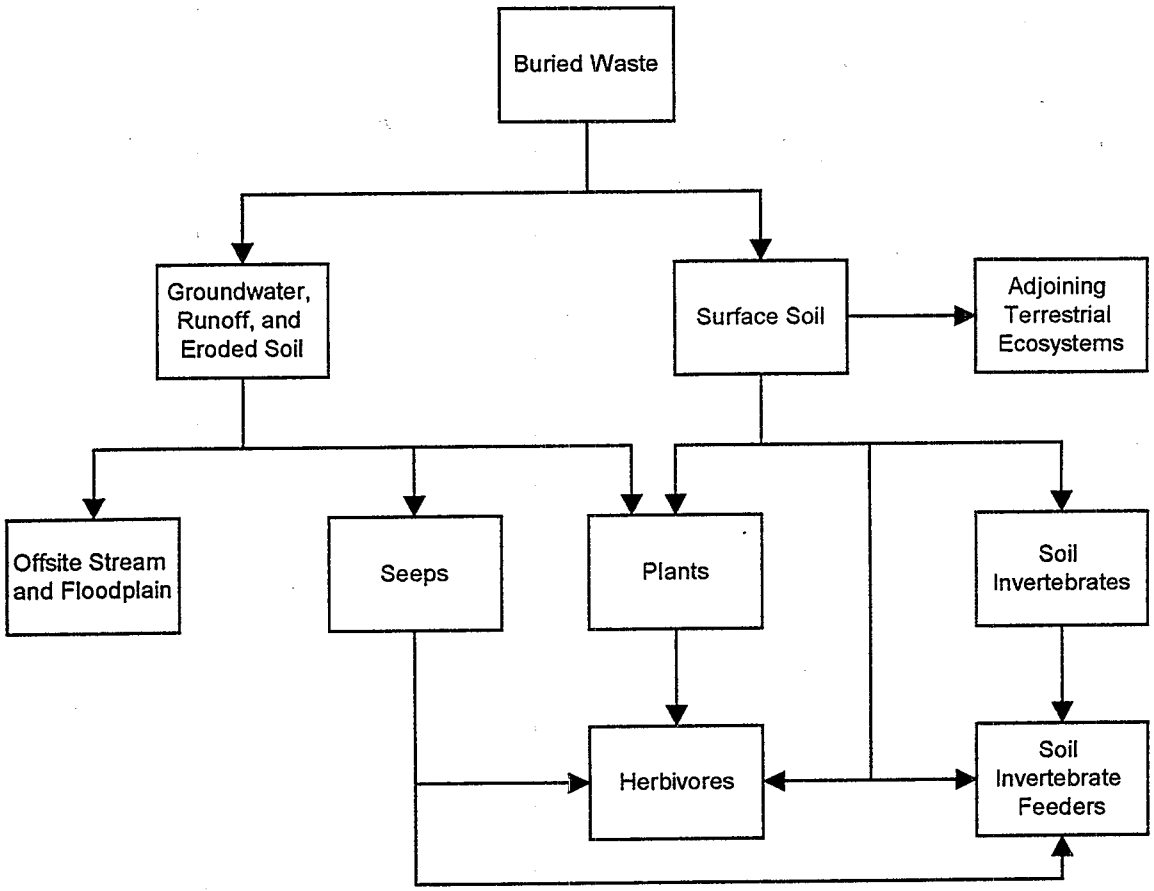

Figure 4-6. A simple conceptual model based on exposure processes for ecological risk assessment of a waste burial ground.

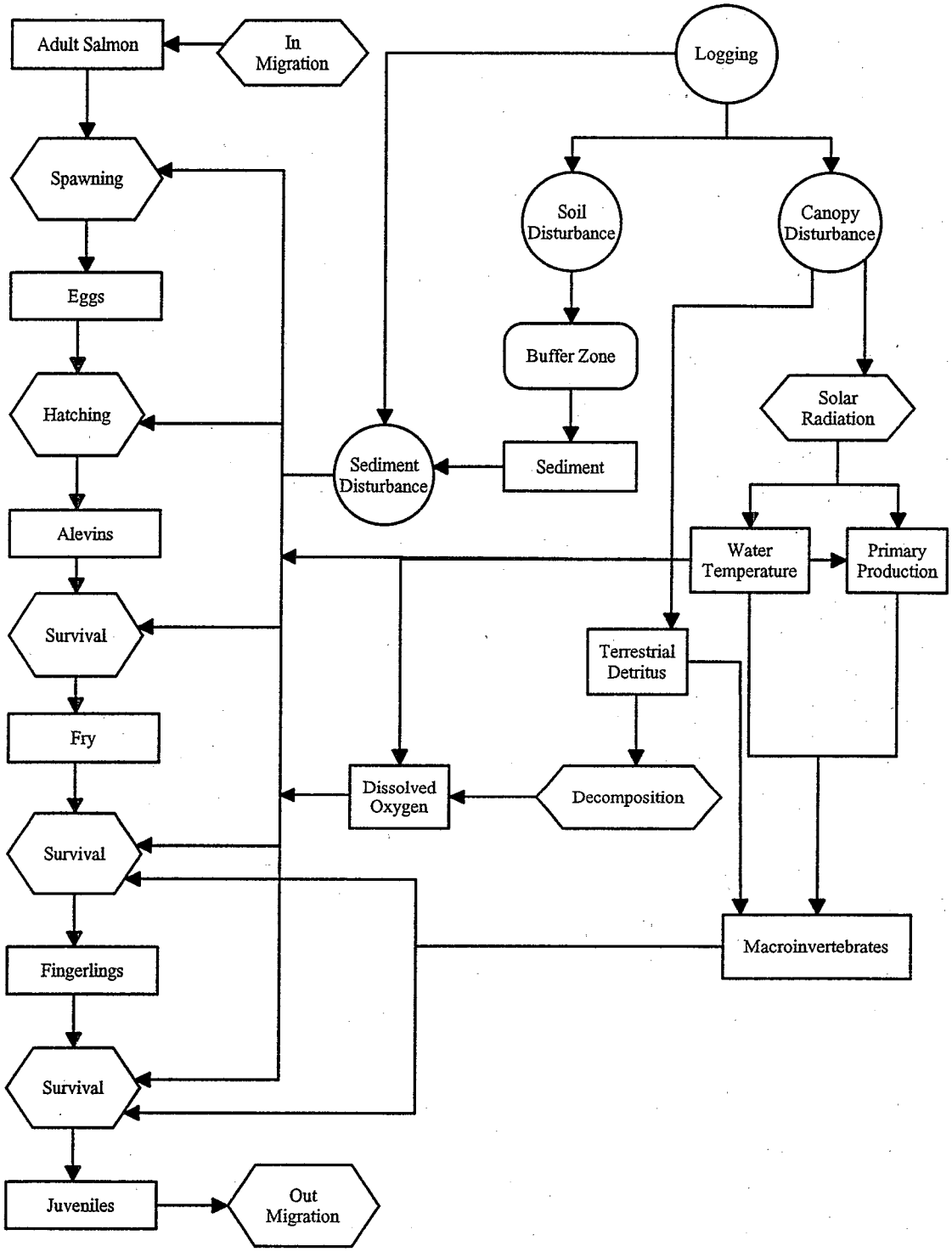

Figure 4-7. A conceptual model for ecological risk assessment of the effects of logging on salmon production in a forest stream. The assessment includes a series of exposures and responses.

3. ANALYSIS PHASE: CHARACTERIZING EXPOSURE FROM ADDITIONS OF PHYSICAL AND CHEMICAL AGENTS

During the analysis phase, the primary duties of the exposure assessor are to (1) further characterize the source and agent, (2) perform a detailed quantitative pathway analysis that includes the formation of secondary agents, and (3) evaluate the exposure process. These three steps are discussed below, followed by a discussion of implementation issues.

Exposure analysis can be quite simple. For example, it can involve quantification of the release, transport, and partitioning of a discrete chemical and its contact with a local population of fish. More complex analyses may involve the transformation of agents and the formation of secondary agents resulting in a variety of interactions with different ecological systems. For example, the release of nutrients in an estuary can increase primary productivity, resulting in increased organic decomposition and eventually decreased DO to the detriment of fish and benthos contacting low-oxygen areas. The characterization of exposure encompasses this entire array of agents, secondary agents, and their contact with ecological components (see figure 4-4).

3.1. Further Source and Agent Characterization

Because detailed knowledge of a source is often essential for an assessment and for any subsequent decision making, it contributes significantly to the ability of the assessor and the manager to address any problems. Sources may be characterized and assessed before operation, either through a permitting process or as a part of liability protection, product stewardship, or life-cycle analysis. Under several laws (e.g., the Clean Air Act, the Clean Water Act, the Federal Insecticide, Fungicide, and Rodenticide Act), new or existing sources are the designated targets of assessment. In retrospective assessments, the objective may be to link a known exposure or effect back to a source so that source management options can be considered.

Sources are commonly classified as mobile (automobile) or stationary (sewage treatment plant); point (pipeline or smokestack) or nonpoint (wheat field, contaminated sediments); deliberate (pesticide application), adventitious (brake fluid leaks), or accidental (spill). In addition, sources are classified by regulations as fully permissible (homeowner's compost pile), specifically permitted by an agency (National Pollutant Discharge Elimination System [NPDES]), or prohibited (banned pesticides).

A complete source characterization includes the specific content, timing, duration, location, and intensity of any releases. The content of the source includes the composition and physical form, the complexity or degree of contamination with known or unknown pollutants or toxicants, and the media

into which releases may occur. The timing and duration of a source may be continuous, intermittent, seasonal, or random or may vary with a prescribed distribution. Location concerns the geographic position of sources as well as its location with respect to transport media (e.g., streams, aquifers) and ecological receptors (e.g., nesting colonies, wetlands). The intensity of the source is the set of dimensions, such as rate of release of a chemical or radiation flux, that determines the magnitude of exposures given a duration and receiving environment.

The definition of the source depends on the scope of the assessment. For example, consider as a typical physical agent the UV spectrum, characterized by intensity at specific wavelengths (as an input spectrum) summed over time. If the effective receptor is known, the exposure (i.e., quantal uptake) could be expressed as the sum over time of incident light energy times the absorption efficiency. The source for an ecological assessment program might be defined as incident radiation given some hypothesized state of the stratosphere; yet for the assessment to lead to actions, it must ultimately address the sources of CFCs. In this case, the exposure assessment would include the transport of CFCs to the stratosphere, subsequent ozone depletion, and modification of the UV radiation as well as radiation absorption (see figure 4-3).

A somewhat more typical agent would be a chemical used as a pesticide. The receptor could be a target organism (pest), a nontarget receptor organism (a beneficial or adventitiously exposed organism), or any other system. The assessment could be linked to a specific or generic site for a particular period (an instance, a program, or the product life) defined by an assessment scenario. Usually product label recommendations specify concentration, formulation, application methods, rate and timing, and crop/target of an application. Through a combination of monitoring, laboratory experimentation, and modeling, the assessor would obtain data on spray deposition to the crop, to soil, and to nontarget organisms as well as concentrations in media attributable to subsequent transport and transformation.

3.2. Detailed Pathway Analysis

A pathway analysis describes the course of an agent as it interacts with the environment. The agent may be transported within a medium, be transferred between media, or be transformed through biotic or abiotic processes, potentially generating secondary agents. The objective is to characterize the spatial and temporal distribution of primary and secondary agents resulting from the addition of agents into the environment.

3.2.1. *Fate and Transport Processes*

Much of exposure assessment for chemical contaminants is concerned with estimating the results of processes occurring within the environment. These can be divided into advective transport and partitioning between media. The other processes that control the fate of a chemical are abiotic and biotic transformations, which often result in formation of secondary contaminants.

The necessary physicochemical and microbiologic parameters for chemodynamic processes have been brought together in a number of databases. Most of these are computer-accessible and include various data quality factors incorporated into their assembly, review, and updating (Howard et al., 1986). In spite of this apparent wealth of data, only a fraction of the values for the estimated 70,000 chemicals in commerce at the time of enactment of the Toxic Substances Control Act (TSCA) in 1976 have been acquired. Experience counsels against extrapolating among chemical classes to fill these data gaps. For example, the bioaccumulation of various substituted aminonitrophenols is unrelated to structure or log K_{ow}, whereas bioaccumulation of the pyrethrins and pyrethroids may vary by more than a factor of 10^5, although log K_{ow} varies by less than 1.2.

3.2.1.1. Advective Transport

The advective transport of an agent with the movement of a medium is dominated primarily by the forces acting on the medium: water moving by gravity, soil blowing in a storm, body residues moving with a bird in migration. Other relevant forces include such processes as solubilization, diffusion, and volatilization that keep the agent in the medium or remove it. Thus a sorbed chemical will move primarily with the soil particle as the particle erodes and becomes sediment to be deposited in the slow-moving reach of a stream. If the particle were then picked up by a filter-feeding invertebrate or otherwise partitioned out of one medium to another, advection by that pathway would be terminated.

Movement within a medium is accompanied by diffusion and suspension/sedimentation (by gravity, convection, turbulence). Intermedia transport—soil erosion, bioturbation, washout of dust from air—also can be a common advective process, but the transfer is mitigated by features different from those in the transport process or by forces acting on the agent in relation to the medium. Thus, whether a soil particle is suspended in air to be blown away by a thunderstorm will depend on the angle and velocity of the impinging drop, the forces binding the particle to the soil matrix (including degree of compaction, soil moisture, organic matter content), the wind velocity, and the temperature affecting density and viscosity of water and air. Similar issues develop in the processes of leaching,

atmospheric rainout, and biomigration. Deterministic processes that are nonlinear and dependent on initial conditions may be deemed chaotic and predictable only by empirical means. Thus prediction of sediment and soil erosion/deposition, leaching, and bioaccumulation within food webs apparently is limited.

Advective transport can span especially long distances. Consider, for example, the transcontinental dispersal of sulfur oxides leading to acid deposition and the contamination of Antarctica and oceanic islands by persistent pesticides. Because clearance of materials from large lakes or aquifers can take centuries, we will need to account for advective transport of bioactive substances for decades at least, even though these materials may no longer be in use. By the same token, an apparently negligible contribution to risk in a site-specific assessment may contribute to a significant risk at a more distant site.

In retrospective assessments, the assessor constructs pathways from a known source and employs the methods described above. Alternatively, the assessor's objective may be to trace the pathways back from a particular location to the original source or sources. Gordon (1988) describes receptor models that combine transport modeling with characteristics of the contaminant mixture at air sampling locations to trace contaminants back through transport pathways to suspected atmospheric sources.

3.2.1.2. Transfers Between Media

Transfers of chemicals and radionuclides between media greatly influence their ultimate distributions. Most exposure analyses estimate the concentrations of chemicals in different media. For simplicity and tractability, these media are assumed to be at equilibrium. In some cases, such as bioconcentration of methyl mercury, equilibrium may never be reached and the analysis would be based on kinetic models and parameters.

The relative equilibrium concentrations of a chemical in two media or phases within a medium is represented as the partition coefficient for water and oil-like substances; the sorption coefficient for soils and sediments in contact with water; and, for such processes as solution/precipitation (solids, liquids) and volatilization/condensation (gases), the solubility product and vapor pressure, respectively. For the last two phenomena, the partitioning only pertains to layers at the interface between media, since mixing of the more distant layers is advective (diffusion, mass flow or circulation). The volatility of a chemical from a wetted surface into air is proportional to its Henry's

Law constant (i.e., the ratio of the vapor pressure to the water solubility at a given temperature and atmospheric pressure).

As noted above, movement of a chemical by partitioning takes place during transport and intermedia transfers. If the equilibrium is very large or very small, such transfers themselves do not displace the chemical, since fully water-soluble chemicals will stay in water and strongly sorbed agents will remain sorbed. In many instances, however, the dynamics of the several interacting processes result in redistribution between media, dispersal and dilution in some instances, and slow and invidious exposure of parts of the environment in others. The rates of the forward and reverse reactions composing the equilibrium constant are assumed to be high enough that interphase transfers take place more quickly than intramedium and intermedia transport.

Partitioning processes are the most thoroughly studied of all chemodynamic processes, and the methods for estimating the equilibrium parameters are well developed, as demonstrated in Lyman et al. (1990). Lipophilicity, as represented by the octanol-water partition coefficient (K_{ow}), is a major factor in environmental chemodynamic processes and is used to estimate water solubility, bioconcentration from water, and soil sorption for nonionized neutral molecules. Thus considerable effort has gone into obtaining sound measurements and robust quantitative structure-activity relationships (QSAR) for K_{ow}. As a result, the relationships of parameters to K_{ow} provide a means of cross-checking measured or estimated values employed in chemodynamic models. Failure of these to check out implies that measurement must be given priority. For example, the natural pyrethrin insecticides and many of their synthetic substitutes have especially high log K_{ow} values but extremely rapid rates of hydrolysis and bio-oxidation; moreover, they are not bioconcentrated as predicted. If the log K_{ow} were relied on as the sole measure of environmental chemodynamics, the bioconcentration of unstable compounds would be overestimated, whereas the uptake and sorption of newly synthesized, more stable pyrethroids would be underestimated.

Partitioning is particularly sensitive to some environmental conditions, especially if the compound has a weakly ionized group that may be affected by pH. Solubility and vapor pressure are particularly sensitive to temperature. For Henry's Law constant, it is critical that both vapor pressure and water solubility be measured at the same temperature (because these features change at different rates as temperatures change). K_{ow} is less sensitive, however, because solubilities in the two solvents usually change at similar rates in the same direction. Computation of the nonionized forms by the Henderson-Hasselbach equation can adjust the distribution for pH but may not properly estimate ion exchange and ionic bonding.

For ionic chemicals, the ion types (i.e., forms with a given valence and degree of substitution, such as Hg^0, Hg^{+2}, CH_3-Hg^{+1}) constitute a *species* into which the chemical is apportioned. Estimating the equilibrium *speciation*—which is dependent on temperature, pH, Eh, and ionic strength—may involve the simultaneous solution of up to 50 equations for as many as 20 species. Empirical knowledge of the binding capacities of different clay minerals and soil and sediment organic matter is required for speciation to be site- and situation-specific. The regularity of reactions of solubility products and complexes, on the other hand, extends speciation as a chemodynamic process in all environments.

The complexity of speciation can be used to illustrate the sorts of practical strategies that are used to estimate ecological exposures. In most cases, assessors begin by assuming that the chemical is present entirely in its most toxic form. If there is no significant risk under this exposure, the complexities are avoided. If significant risks are estimated and if speciation could make a difference in the result, then the assessor must perform a speciation study focusing on the species that are predominant or that contribute significantly to the risk. For some chemicals, such as ammonia, simple and effective speciation models exist; for retrospective assessments, however, it is often possible but expensive to measure species. One simply needs to determine that knowledge of speciation is needed, since specific knowledge of the contaminant is often useful. For example, because mercury in fish flesh is always at least 85 percent methyl mercury, it constitutes the chemical species in the diet of piscivores. Similarly, chromium in soil is trivalent except in unusual circumstances.

Bioaccumulation can be the most important process involving partitioning. Yet because bioaccumulation represents the net effect of uptake, metabolism, storage, and excretion, it is primarily an outcome of partitioning only for metabolically stable chemicals. For fat-soluble chemicals, the bioaccumulation rate is adjusted by the lipid composition of the receiving species. Also, the rate may be very sensitive to species-specific rates of metabolism or excretion. Although there are demonstrable instances in which some chemicals increase in concentration for each trophic level within food webs, this phenomenon of biomagnification does not occur with most chemicals.

3.2.2. *Formation of Secondary Agents*

After its release into the environment, an agent can interact with a wide variety of abiotic and biotic processes, such as chemical oxidation, metabolism, and primary production. The consequences

of these interactions range from complete detoxification of a chemical to chains of events, such as eutrophication, that can completely change an ecosystem's form.

3.2.2.1. Transformation

Because changes in a chemical's structure may activate, inactivate, or detoxify it, information about, for example, biotic and abiotic reactions, the effects of conditions, and requirements for reactants is critical for characterizing exposure.

The role of biotic and abiotic reactions in chemodynamics of environmental pollutants is not always clear. For example, the evaluation of microbial biodegradation is complicated by the role of sorption in preventing degradation (Alexander, 1986). By contrast, surface binding can provide a mechanism for abiotic catalysis of photolysis, hydrolysis, oxidation, or reduction. Counterintuitively, water may inhibit some reactions in which it participates (e.g., soil catalysis of hydrolysis) by outcompeting the chemical for binding sites (e.g., manganese or magnesium). Thus it is usually necessary to clarify the contributions of both biotic and abiotic transformations to a pollutant's fate.

Abiotic reactions. Abiotic reactions are those in which no organism is involved. The nature of many reactions is well known, and the rate may be described by various structure-activity relationships. Oxidation/reduction reactions may be generated by a variety of materials, although the rates are usually 10^2 to 10^3 faster in biological systems. In saturated soils, the creation of anaerobic conditions and a very low Eh determine the biota available for reactions as well as the abiotic consequences. Photolysis is less predictable in terms of rates and products than either hydrolysis or oxidation, but the mechanics may be more accurately described when the quantum yield and light-absorption spectrum are known for a situation in which the incident light can be predicted (e.g., location within the geosphere, time of year). Since a number of products from photolysis are identical or similar to those generated biotically, active agents (e.g., dieldrin from aldrin) may be created by this type of reaction. In some instances, the phototransformation products are unique to that process (e.g., photodieldrin) and may be even more toxic than other abiotic and biotic transformation products. Finally, there is the fairly well-characterized intracellular interaction of photolysis with toxic action (Gala and Giesy, 1992) in which an abiotic transformation process occurs in the receptor organism.

Biotic reactions. Because microbiota make up the majority of life forms and, importantly, are resident in soils and sediments, their ability to transform chemicals is critical to chemical fate. The detailed information controlling microbial action may be in chromosomes or plasmids or may be shared within communities, and it is subject to enhancement or induction under some circumstances. Conversely, microbial transformation may be limited by the lack of previous exposure of the organisms with respect to either transport into the cell or enzymatic transformations thereafter. Thus the chemical history of an area may create an accommodated or acclimated community with respect to a given agent, whereas another microbial community will have had no previous exposure. In the latter, an induction period of up to 2 weeks may be required to achieve competence, after which the biodegradation may proceed quickly (i.e., in a few days). As a result, expressions of biotic stability (or ready degradation) may be misleading when they are based on simplistic calculations of when half of the initial concentration has disappeared.

Microbial communities carry out an amazing variety of reactions. If the chemical can be converted completely to inorganic substances (e.g., carbon dioxide, water, ammonia), the process is termed mineralization. If only one or so steps of degradation are accomplished by a given organism or community—perhaps coincidentally because of substrate similarities between the pollutant and naturally occurring compounds, and without population growth on the pollutant—then the process is termed co-metabolism. Thus this process may be the source of a more persistent moiety or the compound may require transport to a new set of conditions and organisms (e.g., aerobic —> anaerobic) for mineralization to proceed.

Some chemicals that are not mineralized disappear (at least temporarily) from an environment because they are converted to "bound residues"—so called because the chemical cannot be extracted therefrom, although radiolabel studies strongly imply its presence (or that of a metabolite). On the other hand, such labels can result from assimilation of normal metabolites (e.g., conversion of $^{14}CO_2$ to ^{14}C-labeled cell walls) or mechanical entrapment of sorbed materials (e.g., as in a pH shift that closes a clay lattice). Some bound residues may provide a toxicant to biota as the matrix of recalcitrant organic matter to which the binding or covalent bonding has occurred is metabolized. The critical issue regarding bound residues for exposure assessment is to assure the risk manager that a mass balance study accounts for the activity and bioavailability of parent and transformation products over the period of assessment.

Transformations by higher organisms decrease in significance relative to fate and transport of the agent the further one moves from microbiota. Invertebrates and rooted plants may play key roles in

the food web and sometimes are particularly important for the mobilization and bioavailability of chemicals; yet transformations may not be critical. Plants do store some toxicants as conjugates (reacted with simple sugars, amino acids, peptides, and anions such as phosphate or sulfate) that become available to herbivores upon digestion. At the upper levels of the food web, biotransformations are usually important largely in terms of toxicodynamics. Omnivores (e.g., human beings, swine) tend to have broader competence in the range of chemicals handled than herbivores, which tend to have much broader competence than carnivores. In any group, however, biotransformation may be a critical element in selective toxicity (see chapter 5 on effects characterization).

3.2.2.2. Interactions With Ecological Processes

Interactions of agents with ecological processes can result in a causal web that is challenging to trace and difficult or impractical to predict. The complexity of some of these interactions is illustrated by the cascade of exposures and effects that resulted from increasing nutrient inputs into Chesapeake Bay. Complexity increases with the length of the pathway (i.e., the farther in time, space, or transformation steps that the active agent is from the original source or human activity).

A key component of any exposure assessment is the assessor's assurance that the appropriate agent is targeted. For example, the sag in DO associated with the chronic release of biodegradable organic carbon into a waterway can cause mercury in sediments to be methylated under the more anaerobic conditions accompanying that DO sag. Tubificid worms may then proliferate, even as the diversity of mesofauna declines, and acquire tolerated burdens of methyl mercury from the surrounding sediments. Thus these worms mobilize previously unavailable, perhaps even buried, deposits of mercury in the stream. In this case, the exposure assessor would identify mobilized mercury as an active agent to which fish are exposed.

The knowledge of natural processes and the consequences of changes in conditions is clearly a vital part of being able to make such assessments. The ability to anticipate and define these consequences depends on the expertise and experience of the assessment team. This aspect underscores the importance of drawing on soil and aquatic sciences, biogeochemistry, hydrology, and meteorology when conducting exposure assessments. Treatment of these complexities must largely be case-specific rather than generic, so more detailed treatment here is not justified.

3.3. Evaluation and Quantification of the Exposure Process

The exposure process defines the effective contact between the receptors (organism or system) and the contaminated or modified environment. The attributes or behaviors of the receptors that will influence the extent of exposure are evaluated to identify routes of contact with affected media. The degree to which the contact actually results in uptake is determined by the bioavailability of the chemical in the contacted medium. Bioavailability is a function of the properties of the chemical (e.g., speciation), of the medium (e.g., sorptive properties or presence of solvents), and of the biological membrane (e.g., sickness, active uptake). Exposure is quantified through the use of a model, by chemical/physical analyses, or by bioassay. As final steps, the assessor evaluates the role of exposure-based behaviors and other feedback mechanisms in controlling subsequent outcomes.

3.3.1. *Behavioral Attributes*

The time an organism spends in a particular habitat or activity often determines the extent and route of exposure. For this reason, the evaluation of behavior is an explicit part of the exposure analysis of individuals and populations and parallels time-motion studies for occupational health-exposure assessments. Some behaviors such as stomate opening are physiologically controlled such that a particular pathway (vapor phase and particulate intake by plants, in this example) is predictably affected by physical conditions such as drought, heat, and humidity. Animal behaviors, such as feeding, grooming, and digging, determine the dermal, ingestion, and inhalation exposures.

Prey or food selection may include behavioral and physiological mechanisms, but also will be influenced by ecological attributes such as prey abundance and competition. Since animal- feeding studies can reveal behavioral facets as well as body residues and bioaccumulation patterns, exposure assessors should be reluctant to forgo them in favor of methods based solely on chemical structure.

As the receptor system becomes more complex, so does the role of behavior in exposure. Exposure may or may not be a quality of the environment sensed by an animal. In studying the effects of various pesticides on the interactions of crickets (*Acheata domestica*) and voles (*Microtus canicaudus*) in a terrestrial model ecosystem, Gillett et al. (1983) noted that pregnant or lactating female voles (normally herbivorous) ingested significant quantities of crickets, but only if there was foliage present. In open field trials on bare ground, voles and crickets barely seemed to acknowledge each other's existence: Crickets made no effort to avoid voles, and voles never pursued crickets. In the typical ryegrass-alfalfa system used in the studies, however, up to 30 percent of the voles' food intake was crickets.

Where behaviors result in hierarchical ordering of a population (e.g., pecking order, feeding location for a predator), exposures could be graded between individuals because of, for instance, differential access to food or oxygenated water. Failure to obtain enough of an essential resource might subject a receptor organism to disease, predation, or some other adverse consequence lumped into morbidity and mortality.

In addition to directly influencing exposure, behavior in response to exposure can mitigate or enhance exposure. For example, prey that respond more slowly to threats because of toxicity can be subjected to increased exposure to predators (Bildstein and Forsyth, 1979). On the other hand, avoidance of a food item because of an organoleptic response to a toxicant can decrease exposure. These exposure-related behaviors may be considered as effects per se, but they will increase or decrease the presumed direct or indirect exposure of the receptor species. Moreover, although these responses often can be demonstrated in isolation in the laboratory (even as a dose-response relationship), there are few examples of extrapolation to field populations. Should they be known or detected, the assessor must consider ad hoc evaluation of body burden or other appropriate measures of exposure.

3.3.2. *Routes of Contact*

The physical contact of an organism with its environment is limited by one or more membranes. Nonetheless, contact with any one medium may involve multiple routes, which can include inhalation or respiration, ingestion, and dermal transfer. Thus, a trout may ingest water bearing a pollutant, but the majority of contact will be via respiratory transfer (gills) and some dermal transfer, with secondary exposure attributable to food ingestion.

With each route of contact, various factors affect the efficiency of transfer. Routes of exposure that entail low efficiency will be trivial, regardless of the concentration in the medium. Some of these factors for a chemical include its exact form or species, the rate of presentation at the contact surface with respect to the rate of uptake across the surface, and the rate of mixing of the solution or material directly in contact with the organism with the bulk phase of the medium. On the affected part of the organism, contact may be determined by the architecture of the tissue in contact with the medium (which may be a function of age, size, strain, gender, nutritional or physiologic status, or disease condition, as well as temperature), the rate of circulation away from the contact surface and distribution in the organism (which may be controlled by activity or physiologic status), and species- or individually selective features of biotransformation and uptake.

Models of plant uptake are still controversial. Boersma et al. (1991) have proposed that evapotranspiration flux can move chemicals from soil into foliage in a predictable manner within a narrow range of log K_{ow}, but that compounds with higher K_{ow} values are tied up in the roots. Foliar penetration and uptake of gaseous pollutants and particles via the stomata are also important for some chemicals. Because of the difficulties, assessments of plant contamination (and of transfer to parts consumed by animals) are usually evaluated empirically.

Exposure assessment at higher levels of organization (e.g., communities, ecosystems) can be accomplished by estimating the exposure of the component parts or by establishing an operational boundary around the entire unit of interest (e.g., the perimeter of a lake including 30 cm of sediment). For the component-part approach, exposure routes can be evaluated as described above, but attention must be paid to the full distribution of exposure to each component. For the operational-boundary approach, exposure routes can be evaluated as fluxes across the boundary (e.g., atmospheric deposition, inflow, burial in deep sediments, outflow).

3.3.3. *Parameterization of Components*

Once the assessor has identified important routes of exposure, the next step is to qualify parameters influencing exposure. These parameters can include description of the temporal and spatial distribution of the agent, the receptor, the receiving environment, and chemodynamic processes.

Parameters can be measured directly or estimated from empirical relationships. In either case, the assessor should distinguish variability among parameter values in a population from uncertainty in the true value of the parameters. Three measures are generally used to select parameter values: (1) a single value representing a mean or conservative estimate (e.g., upper 95th percentile), (2) a range or distribution of values, or (3) a value specified for a particular scenario. The first approach is commonly used, even though it may yield results that are realistic for only a small fraction of the time and will either underestimate maximum exposures or overestimate population exposures. The third has come to be employed in a handbook-type approach in which an assessor merely fills in some information and calculates a value by rote, which may be appropriate for health protection but which creates a host of difficulties for an ecological exposure assessment. The second approach uses judgments, simulations, or measurements to generate a range and frequency of values, taking into account the statistical properties of the processes and their parameters as a part of evaluation of uncertainties (see chapter 8 on uncertainty in ecological risk assessment).

3.3.3.1. Receptors

Values for many of the receptor parameters of interest are available in the literature for a few birds and mammals (U.S. EPA, 1993). Values for amphibians and reptiles, however, are rare. Allometric equations can be used to estimate values on the basis of body weight when species-specific values are unavailable. Allometric equations are available for food ingestion rates (e.g., Nagy, 1987), water intake rates (e.g., Calder and Braun, 1983), and inhalation rates (e.g., Lasiewski and Calder, 1971; Stahl, 1967).

For mobile organisms, the assessor may need to estimate the amount of time the organism spends in a given area. Data on home-range size can provide a general sense of how far an animal may roam; however, such data provide an incomplete picture because the area in which an animal moves varies with several factors, including reproductive status, season, and habitat quality. Moreover, most animals do not roam or feed randomly within their home range. Further, the term home range has been used inconsistently in the literature and estimates of home range can vary substantially with the measurement technique used. Data on range sizes are generally lacking for fish, although trout have been shown to return consistently to a single rock (Bachman, 1984). Seasonal movements (e.g., anadromous and catadromous species) and even daily migrations (vertical movement of zooplankton) can be important determinants of exposure.

3.3.3.2. Environment

Tabulated historical data are often available on, for instance, stream flows or sedimentation rates or for documenting the geologic, hydrologic, oceanographic, meteorologic, and atmospheric character of the environment. Such measurements permit the creation of subsidiary data in the form of "canonical" or "reference" environments (e.g., a warm, "blackwater" southeastern U.S. stream; a western mountain lake; a southwestern irrigation canal) for which average values have been computed, for example, for the daily insolation spectrum, organic and sediment particle content, hardness, pH, and salinity. Soil maps are available with data on characteristics of both surface and subsurface soils regarding bulk density, organic matter, cation-exchange capacity, pH, water-holding capacity, and crop patterns. These data are of the type characterizing the environment and its conditions affecting chemodynamics.

3.3.3.3. Chemodynamic Processes

A number of empirical methods have been developed and brought together as an alternative to measurement, an aid to planning appropriate measurements, or a check on measurements that have already been made (Lyman et al., 1990). These usually rely on structure or other simple or readily derived properties. Such empirical statistical correlations may have serious errors, putting results in the category of a "good guess." Nonetheless, their use as point values with error terms or as distributions can create a viable picture of the chemodynamics or at least provide a picture that can be verified by laboratory or field studies.

Because data on the properties' receptors relative to chemodynamics and uptake are particularly sparse, a hierarchy of data selection and estimation techniques is required. A measured value of high quality is preferable to a soundly estimated value, which is preferable to data of questionable character. A measured or characterized distribution is preferable to, for instance, a presumed normal, log-normal, arc-sine distribution. Data within a taxon may provide the mean or range of values for unmeasured members of the taxon, but data beyond measured taxa must include all data from taxa at the same level as the range of values. If the distribution of the range of values is unmeasured, it is presumed to be uniform over that range.

3.3.4. *Quantification Methods*

Exposure dimensions must be comparable to effects dimensions. This limits meaningful quantification to three basic sets of techniques: (1) estimation of effective concentrations or doses in media, (2) measurement of residues or biomarkers in the receptors, and (3) bioassays using organisms for which the response-exposure relationship is known.

3.3.4.1. Estimating Effective Concentration or Dose

The calculation of an effective concentration or dose is one of the most common methods used to estimate exposure. Its simplicity is both a strength and a limitation in that, although exposure estimates can be quickly and consistently generated, they can be unrealistic and indifferent to indirect effects (although integrated aquatic exposure-effects models address the latter concern). The approach normally combines modeled or measured concentrations of a contaminant with assumptions or parameters describing contact. For example, exposure to respired media (e.g., water for aquatic organisms, air for terrestrial organisms) is commonly quantified by assuming that contaminants are well mixed and that the organism contacts a representative concentration. For solid media (e.g.,

food, soil) and ingestion of water, ingestion rates are combined with estimated concentrations in dose equations similar to those used for human health assessments.

A simple and effective means of representing individuals or even classes of organisms is via a simple one-compartment model, such as the one employed by Thomann (1981, 1989). Ram and Gillett (1992) were able to assemble a food web of more than 250 vertebrate species and 40 invertebrate "dietary item" guilds, each with a physiologically based pharmacokinetic (PBPK) model (as illustrated in the equation box), such that 90 percent of measured polychlorinated biphenyl (PCB) values for 46 species involved in the site-specific assessment of the model were within the 95 percent confidence interval of the PBPK model outputs. Because the researchers assembled diets for winter and summer seasons—effectively reassembling guilds and dietary items—the number of parameters needed was greater than 10^4. By using a clustering technique to reduce the number of food item guilds and by estimating parameters according to a preset hierarchy, the researchers effectively reduced assembly and computation of the models in scale and scope but not in accuracy of the outputs.

Simple One-Compartment Aquatic Exposure Model

$$
\begin{array}{ccccc}
 & & C_i & & \\
 & & | & & \\
 & & |A,F & & \\
 & k_1 \;|\; k_e & V & & \\
C_w & <==> & C_b & \text{--->} & X_i \\
 & k_2 & & &
\end{array}
$$

where C_w, C_i, and C_b are the chemical concentrations in water, prey organisms, and the receptor; k_1 is the uptake rate; k_2 is the loss rate; A is the absorption efficiency; F is the feeding rate; k_e is the combined elimination rate for metabolism, assimilation, and excretion; and X_i represents the various products, each of which may require a model.

For neutral organics, the ratio k_1/k_2 is directly proportional to K_{ow} and k_e tends to be inversely proportional to K_{ow}, but empirical relationships are available for various organisms and classes of chemicals. For terrestrial animals, a similar representation usually ignores inhalation as insignificant (Ram and Gillett, 1992).

The exposure assessor must consult with the ecological effects assessor to determine the timeframe over which exposure will be averaged (e.g., shorter times for acute and reproductive effects, longer times for chronic effects). As concentrations become more episodic or variable, averaging becomes more problematic. In extreme cases, averaging may not be appropriate at all and assessors may need to use a pharmacodynamic approach.

3.3.4.2. Body Burdens, Bioassays, and Biomarkers

Although most ecological risk assessments express exposure in terms of a measure of external exposure (e.g., concentrations in soil or food), in many cases a measure of internal exposure would be more appropriate (McCarty and Mackay, 1993). The most common type of information concerning internal exposure is body burdens. The chief limitation on the use of these data is the lack of exposure-response relationships in which exposure is appropriately expressed as body burden.

Even when investigators cannot use body burdens to estimate effects, they can use them to help diagnose causes of observed effects in retrospective assessments. Historically, the discovery of an effect led to investigations seeking a chemical, biological, or physical agent responsible for the adverse effect. Such investigations were accomplished by using responses of organisms (bioassay technique) to standard dilutions or similar fractionations. The basis for the response was uncovered iteratively. For example, "chick edema factor" was replaced by measured polychlorinated dioxins and later still by coplanar tetrachlorodibenzodioxins and dibenzofurans. In the case of several avian heart teratomorphs, we now are virtually certain of the causative agents as measured by specialists trained in mass spectrometry. The foregoing process took about four decades, which is an intermediate period between the decade that it took to identify the agents of egg-shell thinning or delayed-type neurotoxicity and the decades spent on the as-yet unidentified causes of conifer decline in Western Europe. It implies that bioassay is still a useful approach that chemical measurement cannot entirely replace.

Quantification of internal exposure may take the form of biomarkers, which are measures of biochemical or physical changes in an exposed organism (Huggett et al., 1992). In general, biomarkers have been even less well related to effects than body burdens. However, they have some potential advantages over body burdens in risk assessment. First, they may be more persistent than a given chemical. An example is the now common use of cholinesterase inhibition as a measure of exposure to organophosphate pesticides. Second, some biomarkers respond to a set of chemicals that have a common mode of action and occur as mixtures in the environment. Such biomarkers may

respond proportionally to the mixture potency, thereby providing a more accurate measure of effects than measurement of the component chemicals. An example of this advantage is the use of the H4IIE bioassay to determine, on a toxicity normalized basis, the exposure of birds to chlorinated diaromatic hydrocarbons (Giesy et al. 1994). In any case, biomarkers are potentially more useful than other measures of exposure because they measure responses that are at or near the action site. Most biomarkers have not proven useful in practice, however, because they are not diagnostic of particular contaminants and they are highly modified by extraneous factors, such as temperature and season.

3.4. Implementation Issues

The discussion below focuses on two implementation issues: the selection and use of models (with particular attention given to validity for the intended purpose) and data acquisition and analysis.

3.4.1. *Selection of Models*

A large number of fate and transport models are available for each medium (for a recent review, see Calamari, 1993) with outputs typically in the form of concentration over time for a particular location. They vary in sophistication, ease of use, applicability to a range of assessment types and approaches, and degree of validation.

3.4.1.1. Verification and Validation

The validity of a model may be in the user's mind and the minds of the audience and is based on review of the scientific principles and assumptions used and on experience in successful application of the model (Suter, 1993). It is necessary to assure peers that one has efficiently achieved a reasonable degree of precision and accuracy within an adequately comprehensive and complete scope. At present, most models in use are computer programs that have been verified for accuracy and dependability in a set of cases. Yet the embodied assumptions may or may not be appropriate, necessary, or sufficient for every application. Indeed, for predictions, most models are limited to a single medium (or to media assumed to be in equilibrium with that medium). Thus there are air and water-plume models that can represent the dispersion from a stack or outfall into an air shed or water body, respectively; there are runoff and leaching models for terrestrial systems; there are multimedia food web models (Ram and Gillett, 1992) and multicompartment models in which the pharmacodynamics within a vertebrate permit modeling of specific organ ingestion (Lindstrom et al., 1974); and there are plant uptake models (Boersma et al., 1991).

With few exceptions, environmental models must have one or more parameters determined from actual data. If those data are from the site or subject area, most would consider the process to be one involving model calibration, which implies limited applicability or transportability of the outcome to other, even similar, sites. For example, biodegradation rates can vary on the basis of the extent to which a site is untainted by previous releases. The value measured may be useful elsewhere with respect to chemical history.

Models may be validated by a reserved data set not used in calibration or otherwise, with validation criteria set forth in advance and justified as to their applicability and reasonableness. Models purporting to accomplish the same task can be compared by the same type of criteria, allowing the user, if desired, to construct a quantitative measure of validity for that task.

Whereas achieving validation of exposure models at the level of environmental organization for a watershed or pond is quite feasible (e.g., Paterson and Schnoor, 1992), we are much less sure of riverine and estuarine systems and of regions or landscapes. Heterogeneity of ecosystems, conditions, and pertinent processes stretches the credibility of "average" results and places excess uncertainty on simulations. There is greater confidence in stream-reach and air-shed models than in watershed and regional models because users are more likely to have obtained data on a consistent basis, perhaps even from one source.

3.4.1.2. Suitability

The exposure-pathway analysis is the chief factor in determining the suitability of any given model for a particular assessment. Once preliminary evaluation has determined the degree of available mass balance or accountability, and calculations have estimated the relative proportions to be allocated to each pathway, the assessor must examine the models for these media in the pathway(s) to ascertain whether assumptions and available data for parameterization are suitable.

A model also must be compatible with the ecosystem and with the temporal and spatial dimensions of the assessment. For example, advective transport of soil and soil-associated chemicals is most commonly estimated using the Universal Soil Loss Equation (USLE), which was intended to represent long-term impacts of average rainfall with agronomic use with a view to selection of soil conservation and management techniques (Wischmeier and Smith, 1978). Attempts to use this model to represent transfer of sorbed material from unmanaged soils will likely not yield an accurate appraisal if empirical data are used within too narrow a span of time (Risse et al., 1993). Alternatives include SWRRB (Arnold et al., 1990) and AGNAPS (Young et al., 1987).

3.4.2. *Acquisition of Adequate Data*

The assessor must plan data acquisition so as to provide sufficient data of adequate quality to produce a risk estimate that has acceptable uncertainty for purposes of risk management. In the past, no small part of the uncertainty could be attributed to data that were inadequately obtained, managed, and evaluated, then unceremoniously assembled or passed on to the manager. With the advent of data quality objectives (DQOs) and good laboratory practices (GLPs) and establishment of quality assurance/quality control (QA/QC) plans as standard operating procedures in testing and analysis, concern for data quality and adequacy has become standard for exposure-assessment professionals. Although not every instance or assessment demonstrates these principles, they are at least firmly established in theory and practice. Estimation of measurement error and analysis of sampling frequency and density will determine the adequacy of any particular set of measurements (Gilbert, 1987). There is a constant battle, however, between the sufficiency of sampling and costs of analysis. Guidance in sampling needs to take cost effectiveness into account via a parsimonious approach (e.g., Boomer et al., 1985).

Simply put, the data must be robust enough to support management's decisions. Background and controlled exposures must be described sufficiently to permit characterization of the causes of measured exposures. Data acquisition must be carried out by approved and tested methods, with trained personnel using well-maintained and standardized equipment. Data management must provide for identification of where, when, by whom, and how each sample was made or taken—for the surety of data storage and sample archives; for the verity of the data itself; and for correction, should errors occur. For predictive assessments, the data quality objectives are relevant to the parameterization of models, validation studies, and calibration to field sites.

Data adequacy for exposure characterization also is constrained by time limitations imposed by inputs from those responsible for characterizing the ecosystem and those with the responsibility for integrating exposure and response assessments: (1) the time factor of the response with respect to the rate of change or variation of the agent, and (2) the time to decision. Clearly, if the manager must reach a decision in the near term, sampling intensity and analytical methodology will be subjected to different forces than when the decision is not time sensitive. This time-to-decision factor may be determined by, for example, the interval between breeding seasons for an endangered species, the time it takes for a critical concentration to build up in estuarine sediments, or a budgetary imperative; in other words, anything theoretical or pragmatical that can limit or force a decision, no matter how prosaic.

3.5. Summary

The objective of the exposure characterization is to provide a profile in time and space of the distribution or intensity of exposure to the agent and its products. For chemicals, this requires (1) characterization of the source, (2) characterization of fate, transformation, and transport, and (3) characterization of the interactions of the organism and contaminated media. This requires a combination of measurement and modeling, but the relative contributions of those activities depends on the nature of the assessment. In any case, the quality of the exposure characterization depend on scrupulous attention to the adequacy and quality of both the measurements and the models used.

4. THE ANALYSIS PHASE: CHARACTERIZING DISTURBANCES

4.1. Direct Disturbances

In general, analysis of direct exposure to disturbances such as logging activity or the construction of dams or parking lots is a relatively simple process once the problem formulation stage of the assessment has specified the appropriate units to be used for measuring intensity, time, and extent of exposure (section 2). Unlike exposure to chemical, biological, and physical agents, assessments of these types of disturbances require no modeling of exposure pathways or measurement of contaminant concentrations, biomarkers, or radiation levels—the wetland is filled, the fish are harvested, the valley is flooded.

Thus the analytical phase for such assessments consists of specifying the particular attributes of the disturbance that are of concern and then quantifying the effective deletion or modification. Often the process involves relating the area modified to the areas occupied or used in particular ways by the endpoint populations or communities. For retrospective assessments, such refinements can be based on observations and measurements of both resources and disturbances. For predictive assessments, the location of resources is specified in relation to the proposed disturbances. This task has been simplified by the availability of technologies such as global positioning systems, optical scanners, image analyzers, geographic information systems, and relatively inexpensive computers.

The process of overlaying the distribution of the disturbance onto the distribution of the valued resources to establish the assessment endpoints can be considerably improved by defining the patterns of disturbance for more recent as well as all previous disturbances within the range of the resource. For example, the habitat of a species may be defined as a set of habitat patches that supports populations via corridors that permit genetic exchange and repopulation. An individual disturbance that removes a relatively small portion of the total habitat may sever a corridor or render a habitat patch too small to sustain a population. Analogous large-scale spatial considerations also could enter into assessments of the functional properties of ecosystems and regions. Such an approach borrows from the relatively new fields of landscape ecology and conservation biology.

The temporal scale of the disturbance also may be easily defined. If the disturbance is transient (e.g., logging a site or clearing it by burning the vegetation), the temporal scale of the risk is determined by the recovery process, which is logically part of the effects assessment, and by the recurrence frequency. If the disturbance continues, the duration of the disturbance and of recovery must be considered. For example, a reservoir disturbs a stream and riparian ecosystems until it "silts-in" or is removed. In some cases, however, recovery is not an issue; for example, ecosystems

affected by the construction of highways or industrial sites are seldom allowed to recover. Similarly, implementation of a resource management plan removes fish, wildlife, timber, or forage for a finite period, after which the plan is replaced by another.

Although most of these direct disturbances are essentially of an all-or-none type, some have gradations of intensity that the assessor must specify. Harvesting may remove variable numbers of organisms of varying properties (e.g., age, sex, size) and some system modifications may vary in intensity, such as the depth to which a wetland is drained.

For these direct disturbances of the environment, the assessment burden is largely on the effects assessor, who must translate harvesting or exposure to the modified environment into effects on the assessment endpoints (see chapter 5 on effects characterization).

4.2. Secondary or Indirect Exposures From Disturbance

Although most effects of disturbances are experienced by the ecosystems that are directly disturbed (e.g., the forest logged or the woodland developed for industry), indirect disturbances also may be significant (e.g., the siltation of streams by agricultural tillage, logging, strip mining, and construction). Thus indirect disturbances should be considered in the problem formulation phase of the assessment.

A sometimes challenging part of the analysis phase, however, involves identifying the specific consequences of the disturbance that will affect the assessment endpoint. For example, the removal of riparian vegetation can generate multiple secondary disturbances; yet it is the resulting increase in stream temperature that appears to be the primary cause of adult salmon mortality. Different secondary disturbances influence other endpoints, such as decreased spawning and egg mortality.

Methods appropriate for analysis of indirect disturbances are specific to the type of disturbance. Often, assessors follow potentially useful empirical and theoretical/mechanistic approaches. For example, estimates of erosion and subsequent siltation may be based on experience with a particular type of disturbance (e.g., strip mining in the region) or on models (e.g., Arnold et al., 1990; Young et al., 1987). In general, if the process by which indirect disturbances occur is complex, and if in the past similar disturbances have been studied, empirical data will provide the best evidence for the assessment of proposed disturbances (Goodman, 1976).

5. PRESENTATION OF RESULTS: THE EXPOSURE PROFILE

The product of the exposure characterization is an exposure profile that is important for the risk characterization phase of the risk assessment. This section discusses the need to express exposure in appropriate dimensions, to acknowledge and quantify uncertainty, to describe the methods and assumptions used, to provide tiers of exposure characterization, and to note how this step relates to broader aspects of risk management.

5.1. Dimensions of Exposure

For the results of an exposure or disturbance assessment to be useful, they must be expressed in units that correspond to the exposure or disturbance dimensions that determine the risk. In particular, the units of exposure must be commensurate with the units of the exposure-response relationship generated by the effects assessment if they are to be integrated in the risk characterization.

In general, exposure has three dimensions: intensity, time, and extent. Intensity is expressed as the concentration or dose for chemicals, the dose for ionizing radiation, and the level of nonionizing radiation. Analogous measures of intensity can be identified for other agents. For changes in hydrology, level or stage is an appropriate measure of the intensity of riparian flooding (Brody et al., 1989; Pearlstein et al., 1985), wetland status (Lyon, 1993), or aquatic habitat (Orth, 1987). Harvesting intensity is the rate of removal or the number or mass removed. For habitat destruction or modification, intensity may be expressed as changes in the levels of habitat characteristics (e.g., primary production or foliage height diversity) that determine density or productivity of the endpoint populations or communities (Bovee and Zuboy, 1988).

The temporal dimension of exposure is usually expressed as duration (Suter, 1993). It may be expressed simply as the time over which exposure occurs or exceeds some threshold intensity or as a variable over which intensity is integrated. If exposure occurs as repeated, discrete events without significant variation in duration (e.g., logging or discrete chemical spills), the frequency of recurrence is the important temporal dimension of exposure. If the repeated events have significant and variable durations, then both types of temporal dimensions must be considered. In cases where the seasonal periodicity of an exposure influences the extent and magnitude of effects (e.g., influx of hydrogen ions and aluminum during snow melt), this factor should be described in addition to the duration and frequency.

Extent has seldom been incorporated explicitly in ecological risk assessments, perhaps because the assessments tend to focus on the estimation of effects at the point of maximum exposure. When it

has been considered, it usually has been defined simply as the area exposed or disturbed. It also might be the area over which exposure exceeds some threshold of intensity or duration, the areas within which particular categories of intensity or duration occur, or a variable over which intensity and duration are integrated. At larger spatial scales, however, area alone may not be an adequate descriptor of extent of exposure. The effective extent may be a function of the pattern of exposure, including the number of areas and their shape and arrangement. A general solution to the problem of incorporating pattern into ecological assessments has yet to be developed, but landscape ecology provides concepts that assessors can apply in individual cases.

Assessors can treat the dimensions of exposure in various ways: as variables, as constants (i.e., by collapsing to a single point), as factors to be ignored, or as new variables created through combining (e.g., the use of the product of concentration and time as a dose variable) (Suter, 1993). In any case, guidance for ecological risk assessment should insist that the assessor states how each of the three general dimensions is being treated and why that treatment is necessary or appropriate.

5.2. Uncertainty

Characterization of exposure, more than characterization of effects, lends itself to formal quantitative uncertainty analysis. Monte Carlo simulation and similar error propagation techniques are applied easily to the mathematical models used to predict or describe transport and fate of contaminants. Many of the environmental parameters that force the use of these models (e.g., wind speed and direction, flow rates, precipitation rates) are stochastic, and their distributions can be specified from weather records, hydrologic records, and similar data sets. Many other parameters are known without error (e.g., molecular weight, activity of radionuclides) or their uncertainties can be reasonably bounded (e.g., K_{ow}, solubility). Therefore, assessors who model exposures to additions of chemical and physical agents should be taking the lead in applying quantitative uncertainty analysis to ecological risk assessment. The fact that exposure assessment is applicable to uncertainty analysis suggests that it may be appropriate to present exposure uncertainties in terms of the probabilities of exceeding exposure levels estimated to be thresholds for significant effects. Nonetheless, certain other sources and expressions of uncertainty also would need to be presented (Funtowicz and Ravetz, 1991). (For more on uncertainty, see chapter 8.)

5.3. Iteration and Level of Detail

The exposure profile may be a "work in progress," as new information, process understanding, and analysis techniques are applied within the problem formulation, effects assessment, and risk characterization steps. This naturally means that iterations of the exposure profile will have increased complexity and analysis will cost more. It therefore would seem useful to consider when and how the exposure profile might be simplified, as in the early tiers of an assessment, and when the more detailed and costly effort is worthwhile and advisable. Alternatively, the concept of a tiered exposure profile could also represent the rationale for omitting some steps or analyses, for example, of an exposure pathway not likely to be appropriate or significant in the final risk characterization.

In part, the methods and models appropriate to tiered analysis are among the suite of steps used to assess preliminary, incomplete, arbitrary (default), or inadequate data, such as the prebiologic screen for ecotoxicological effects (Gillett, 1983) or related efforts. For example, if the concentration of carbofuran in a granule would require that a bird species consume several days' ration to acquire enough material to achieve 10 percent of an LD_{50}, and if the half-life of the carbofuran in the granule were only a fraction of a day, then postulation of an ingestion pathway to that bird species (which might require additional data gathering to complete a more detailed exposure profile) might be forgone as not worthwhile.

The tiering of the exposure-assessment products might, therefore, take the form of (a) brief screening assessments justifying eliminating or disregarding certain agents, pathways, or receptors; (b) more complete analyses that estimate exposure for those agents, pathways, and exposures that pass the screen; and (c) detailed analyses focusing on reducing uncertainties identified in the previous tier. This should not be taken as a recommendation that all exposure assessments be performed in three tiers. One should add tiers until the exposure profile is sufficient for a risk-management decision. Some of these steps will be in the minds of the problem formulation team at the start of the exposure assessment, in anticipation of choices to be made operationally within that assessment. Initially, some steps will derive from iterations of the full assessment and comparison of model outputs to field data. Any factors promoting stability of the agent in the environment, broadening the circle of affected species and individuals (as consumers, say, of poisoned prey), or increasing the duration or extent of exposure beyond the site would trigger another iteration of the basic exposure profile in a more detailed form.

When complete, the tiered set of exposure profiles of a given activity or event would likely not be the same for all portions of the environment, but rather should reflect the expert judgment of

assessors about all significant potential routes of exposure and analysis of uncertainty. This complete set would be adequate to describe all pertinent questions needed for assessing impacts within the receiving environment. This might include, for example, ancillary pathways generated by metabolic products having different modes of action (not additive with that of primary agent) or the quantification of risk cascades anticipated from imposition of a physical agent or deletion of some component.

In effects-driven retrospective assessments, the exposure profile is the primary (but not the only) item of interest, although one might also ask why certain effects did or did not occur from historical releases or activities. In this instance, there is less need for a three-tiered system, inasmuch as pertinent data may no longer be recoverable. The first tier of assessment would qualitatively analyze the value of pathway data, while the next iteration would provide pertinent details, if necessary.

5.4. Relationship to Risk Management

Ecological risk assessments are usually driven by legal, economic, or other management goals and perceptions (e.g., the requirements of the Endangered Species Act and the Federal Insecticide, Fungicide, and Rodenticide Act, as described in chapter 10). The product of exposure characterization must fit the needs of such management decisions. Indeed, it is often the exposure—rather than some mitigation or remediation of effects—that is eliminated or altered as a result of the assessment.

6. GAPS IN KNOWLEDGE

In our view, the following data gaps are of particular importance for characterizing exposure in ecological risk assessment:

1. With respect to exposure to chemicals, additional information is needed, primarily on bioavailability. EPA is addressing issues concerning the bioavailability of chemicals in sediments and is beginning to address issues with respect to metals in water, where the problems are most conspicuous. Yet we need to consider bioavailability for all contaminants and routes of exposure and to address numerous questions. For example, how available are organic chemicals that are sorbed to suspended and dissolved organic matter? How available are compounds in various soils to plants and animals? Moreover, we need to undertake research to allow prediction and measurement of bioavailable forms.

Bioavailability issues encompass more than merely sorption/desorption considerations (which are still an issue in nonaqueous phase liquids). For example, the route of uptake in relation to residue stability (bioconcentration vs. bioaccumulation) is a vexing problem; covalent bond formation to soil and sediment organic matter followed by ingestion or biodegradation may release the agent or a transformation product; enterohepatic cycling exposes chemical agents to microbial and organismic modification. Some of these are steps to complete mineralization and detoxication; in other circumstances, the result is activation.

A consequence of the uncertainty surrounding bioavailability in the terrestrial environment is the current inability to reliably predict chemical concentrations for even low trophic levels. Research is needed to improve accuracy and precision in estimates of concentrations in plants, amphibians and reptiles, insects and small mammals, and birds.

2. Decision rules or other direction is needed for determining what chemical characteristics and environmental circumstances to consider for various modes of exposure. For example, some uptake routes, such as inhalation and dermal absorption, are seldom considered in wildlife-exposure assessments, although any one of these routes could exceed the conventionally estimated dietary exposures in particular cases. Similarly, recent investigations of trout at the Milltown Superfund site found dietary uptake to be a significant route of exposure to arsenic, cadmium, copper, lead, and zinc. This suggests that we need to reconsider dietary exposure to chemicals in aquatic systems. Finally, when nitroaromatics were investigated, there was an implication that they or their metabolites were transpired from the leaves, thus laying munition residues open to mobilization by a route that has not been thoroughly explored.

3. Although the modeling of transport and fate of chemicals is generally the best- developed component of ecological risk assessment, the estimation of biodegradation is an exception. Better ways to estimate the biodegradation of chemicals in various media and conditions are needed. Whether the transformation results in increased or decreased toxicity, for example, is of less importance to the exposure assessment per se than whether it changes the exposure pathway and critical exposure concentrations at various points with respect to receptors.

4. The translation of areas physically disturbed by an individual action into appropriate units of exposure for assessment of the viability of populations requires a better understanding of regional population ecology. Currently, the favored paradigm for assessing terrestrial animals uses habitat units connected by habitat corridors. The utility of this paradigm is unknown, however, and no generally accepted approach for determining whether a particular disturbance will effectively eliminate a habitat unit or corridor has been developed.

5. Better data and methods are needed for developing conventions for averaging concentration measurements or other exposure estimates for aquatic and terrestrial organisms and populations. While range size currently is used to establish areas for developing spatially averaged chemical concentrations, the method is ineffective for animals that are especially mobile or for those that are particularly selective in their use of microhabitats. The issue is aggravated in situations where concentrations are highly variable, such as with soils at hazardous waste sites. Similar issues arise in temporal averaging of aqueous exposures.

6. A collective examination of disturbances as a class might lead to useful generalizations about how exposures to disturbances can be most effectively described. The existing literature in disturbance ecology should be helpful, including Connell (1979), Levin and Paine (1974), Picket and White (1985), Veblen (1992), and Woodley et al. (1993).

7. Although internal dose is usually a better estimator of exposure than external dose, there is currently little basis for using internal dose metrics in ecological risk assessment. While this is primarily an issue of concern to exposure assessors, the information gap is on the effects side. That is, we know how to measure body burdens of contaminants and biomarkers, but we have few exposure-response relationships based on body burdens or biomarkers.

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GLOSSARY

Note: Most of the exposure-related terms defined in this glossary are not defined in the *Framework Report* (U.S. EPA, 1992a). Others propose expansions or refinements to those in *Framework Report*. The exception is the term agent, which is preferable to the term stressor, which implies a prejudgment of the results of the assessment and which is inconsistent with other EPA risk assessment guidance. Most of the problematical terms in ecological risk assessment are generic terms used heavily in documents such as frameworks and guidelines. They are much less problematical in ecorisk practice because they are replaced by more specific terms (e.g., dredging or diazinon rather than agent or stressor).

Agent. An entity that is released from a source or is used to conduct a human action, or the human action itself. This entity is considered the agent in a risk assessment. An agent may affect a receptor in a beneficial, neutral, or injurious manner. Agents may include chemicals, dams, fishing trawlers, exotic organisms, or suspended sediments. The term is used in EPA's *Guidelines for Exposure Assessment* (U.S. EPA, 1992b) (i.e., with exposure defined as "contact of a chemical, physical, or biological agent"), is defined there for human health assessments, and is applicable for ecological risk assessment with the following modification:

> *An agent is a physical, chemical, or biological entity that is released to or imposed upon the environment from a source.*

This definition covers the range from individual chemicals to fishing. It is equivalent to the term stressor (see below) but is used here for consistency with conventional risk assessment terminology and for its neutral connotation in common English usage.

Bioavailability. Chemicals in a form such that if organisms were in contact with it, absorption could occur, are bioavailable. Bioavailability is specific to the type of organism, chemical, and route of exposure. An appropriate definition is found in the EPA's *Guidelines for Exposure Assessment* (U.S. EPA, 1992b):

> *The state of being capable of being absorbed and available to interact with the metabolic processes of an organism.*

EPA goes on to explain: "Bioavailability is typically a function of chemical properties, physical state of the material to which an organism is exposed, and the ability of individual organisms to physiologically take up the chemical."

Chemodynamics. In parallel with the term pharmacodynamics, long used to refer to the movement of chemicals in the body, chemodynamics can be defined as:

The set of physicochemical, advective, biotic and mechanical processes operating on a chemical or other agent that determine its spatial and temporal distribution in the environment.

Chemicals flowing through an organism or moved in an abiotic medium (air, water, soil) may have effects even in that process, and those effects may or may not be the subject of the direct assessment. Thus bioaccumulation may involve any number of organisms in several tiers of the food web before it may be involved with the object of the effects assessment. All of those steps are part of the chemodynamics as far as that receptor would be concerned. Yet the pharmacodynamics in each organism may be important to the overall effects assessment as well as the concentration of the agent resulting within a given scenario. The ambiguous nature of this perspective can be confusing to assessors and managers.

Contact. EPA defines exposure in terms of "contact . . . with the outer boundary of an organism" (U.S. EPA, 1992b). This is appropriate for assessments performed at the organism level and the population level, because assessments of populations are performed on the basis of the average exposure or some other integral of the exposures of the component individuals. Yet it is often appropriate to speak of the exposure of an ecosystem as a unit rather than in terms of the component exposures. Examples include exposure of a terrestrial ecosystem to atmospheric deposition or of a lake or river to inputs from its tributaries. Thus, an appropriate definition is:

Contact is the occurrence of an agent at an interface with a receptor.

Co-occurrence. In the definition of exposure, the term co-occurrence is added to account for exposures that are not a result of direct contact, including competition by an introduced organism and reduced dissolved oxygen due to organic pollution. Thus, an appropriate definition is:

Co-occurrence is the occurrence of an agent in sufficient spatial and temporal proximity to a receptor to result in a response if the magnitude of exposure is sufficient.

The concept of sufficient proximity depends on the dynamics of the agent, the receiving environment, and the endpoint entity. For example, proximity of fish to an organic pollution source is a function of flow, degradation rate of the pollutants, oxygenation rates, and tolerance of the fish for low DO. It must be emphasized that the concept of co-occurrence does not apply to agents such as chemicals that require contact for exposure to occur.

Disturbance. This term is applied to physical deletions or modifications, including logging, dredging, and flooding. The definition is a slight modification of the one provided in a seminal paper in disturbance ecology (White and Picket, 1985):

Any event or series of events that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment.

Defined in this way, disturbance is clearly a kind of exposure (i.e., an event that subjects a receptor, the disturbed system, to the actions of an agent).

Dose. Definitions of dose are legion. EPA uses approximately 140 words to define the term with regard to human exposures to chemicals (U.S. EPA, 1992b). EPA provides a general definition and then defines five varieties of dose (some of which do not fit the general definition), but does not cover radiological dose to humans or ecological uses of the term. For example, air pollution phytopathologists commonly define dose as the product of concentration and time (ppmh) (McLaughlin and Taylor, 1985). A general definition, adapted from Suter (1993), is:

Dose is an integral measure of effective exposure.

Any attempt to define dose more narrowly for ecological risk assessment is likely to fail because diverse uses are deeply embedded in the sciences involved in exposure assessment. When the term dose is used, the user must be careful to define it in context. Dose is used for chemicals, radiation, and pathogens; there is no need to extend the concept to other agents.

Exposure. The term exposure is as appropriate to ecological risk assessment as to human health risk assessment. Fish and trees can be exposed as well as humans, and populations or ecosystems can be exposed as well as individuals. The term does not apply only to chemicals. Organisms are commonly said to be exposed to radiation, pathogens, or heat. The term can also be applied to other agents, such as exposure of a benthic community to dredging, exposure of an owl population to habitat modification, or exposure of a wildlife population to hunting. If these uses seem stilted, the alternative term *disturbance* may be used.

Although the operational definition of exposure, particularly the units of measure, depend on the agent and receptor, the following general definition can be used:

Exposure is the intensity, distributed over space and time, of contact or co-occurrence of an agent with a receptor.

This definition is equivalent to EPA's definitions: "contact of a chemical, physical or biological agent with the outer boundary of an organism" (U.S. EPA, 1992b) and "co-occurrence of or contact between a stressor and an ecological component" (U.S. EPA, 1992a), except for (1) emphasis on the intensity of the agent (e.g., concentration, dose, abundance), (2) inclusion of spatial and temporal dynamics, and (3) the idea that for some agents, such as an introduced competitor species, co-occurrence is a more appropriate concept than contact.

Note that exposure also may be defined as a process rather than an event. The definition in Suter (1993) can be expanded as follows:

Exposure is the process by which an agent comes into contact with or co-occurs with a receptor.

Exposure pathway. EPA's definition (U.S. EPA, 1992b) can be adapted as follows to apply to chemical, physical, and biological agents that act directly on a receptor:

The exposure pathway is the physical course a chemical, physical, or biological agent takes from the source to the receptor.

The term must be adapted further to be applicable to environmental deletions and modifications:

The exposure pathway is the physical course a chemical, physical, or biological agent takes from the source to the receptor or is the causal pathway by which an agent directly modifies the receiving environment, produces any secondary modifications, and exposes a receptor to an ultimately modified environment.

The term is not applicable to the processes by which indirect effects are induced; the terms *causal pathway* or *influence pathway* should be used for such processes or for the combination of the exposure pathway and the pathway by which indirect effects are induced.

Incremental exposure. An appropriate definition is:

Incremental exposure is that component of exposure that is attributable to the source being assessed; it is in addition to background exposure and exposure attributable to other definable sources.

Intensity. An appropriate definition is:

Intensity is that dimension of exposure that serves to indicate the capacity of a particular agent to affect a receptor given a constant duration and extent of exposure. Examples include concentration, dose, and harvesting rate.

Interval/frequency. The recurrence of episodic exposures can be defined as the frequency of occurrence or the length of the interval between recurrences. It may be defined as the mean, minimum, maximum, or planned exposure, or any other expression of a temporal variable that may be used.

Receptor. An appropriate definition is:

The receptor is the biological system that is exposed to the agent.

Receptor is a convenient, general term that is clear in most contexts and is applicable generally to tissues, organisms, populations, and ecosystems. "Ecological component" is the term in the

framework for the things that are exposed; Cohrssen and Covello (1989) use "biological system," which is more general. Either of these is acceptable and may be clearer in some contexts; however, "receptor" is usually clearer in discussions of exposure where the emphasis is on the source-receptor relationship.

Source. An appropriate definition is:

An entity or action that releases to the environment or imposes on the environment a chemical, physical, or biological agent or agents.

Thus sources may include a waste treatment plant, a pesticide application, a logging plan, a release of a novel organism, or a dredging project. In general, it is more useful to define sources in terms of the action that releases the agent rather than simply the entity that is used to release it (e.g., the spraying rather than the sprayer). The recently popular term "management alternative" seems to be a hypothetical management plan and would constitute a source. Depending on the scope of the assessment, a source may be a proximate source, such as an application of a pesticide, or an ultimate source, such as registration of a pesticide for a particular use.

Stressor. This term has been redefined and broadened by the Risk Assessment Forum to include both agents and the primary effects of an agent that can cause secondary effects. The new definition is:

Any agent that can induce an adverse response. The stressor may be an agent or may be derived from interactions between an agent and the ecological system.

Stress regime. The term stress regime has been used in at least three distinct ways. (1) It has been used by some environmental toxicologists to indicate the exposure to multiple chemicals or to both chemicals and physical agents such as heat. (2) It was proposed at one of the issue paper workshops that stress regime be used as a preferred alternative to exposure because it is less suggestive of chemicals. (3) It was used by at least one issue paper author to indicate the series of interactions of exposures and effects resulting in secondary exposures, secondary effects, and, finally, ultimate effects. Because these distinct definitions have been proposed by issue paper authors and

reviewers, there is considerable opportunity for confusion if the term is used. Moreover, there are clearer terms for each of these three concepts. The concept in the first definition is usually referred to as multiple exposure, complex exposure, or exposure to mixtures. The concept in the second definition is exposure. The concept in the third definition has been referred to as a causal chain, causal pathway, or causal network (Andrewartha and Birch, 1984), or risk cascade (Lipton et al., 1993). Because of the potential for confusion and the availability of terms that more clearly convey the concepts to most readers, we have avoided the use of this term in this issue paper.

Susceptibility. Susceptibility is commonly used to indicate the characteristics of an organism that determine its magnitude of response to a particular exposure (i.e., as an antonym to *resistance*). EPA (1992a) has used it to indicate the characteristics of an organism that determine its magnitude of response to particular ambient contamination levels and distributions (U.S. EPA, 1992a). In other words, EPA includes characteristics of an organism that determine its exposure as well as its sensitivity to the exposure. Although this use is somewhat unconventional, we adopt it here because there is no generally accepted alternative term for this important concept. (One possible alternative is *vulnerability*.) Thus an appropriate definition is:

The characteristic of an organism or other system that determines the magnitude of its response to a contamination or disturbance of the environment.

Total exposure. An appropriate definition is:

Total exposure is exposure to an agent resulting from all sources and pathways; total exposure equals incremental exposure plus background exposure plus exposure to sources or pathways outside the scope of the assessment.

This definition can be applied to actions as well as entities (e.g., total exposure of wetlands in a region to dredging and filling, where each permitted action is a source), or the term total disturbance may be substituted. The definition also could be used for a class of agents having the same mode of action (e.g., entrainment of larval fish in hydroelectric turbines, cooling systems, and intakes of water treatment plants).

Issue Paper
on

EFFECTS CHARACTERIZATION

Patrick J. Sheehan
ChemRisk Division
McLaren/Hart
Alameda, California

Orie L. Loucks
Department of Zoology
Miami University
Oxford, Ohio

Prepared for:

Risk Assessment Forum
U.S. Environmental Protection Agency

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1. INTRODUCTION

Characterization of the direct and indirect effects potentially induced by exposure of an ecological system to chemicals or to treatments relating to resource harvests is one of the larger components of risk assessment (see chapter 1 for a discussion of the framework). How such characterizations are approached depends substantially on the conceptual model for ecological risk assessment (see chapter 3). In this paper, we seek to complement the discussion of exposure (chapter 4) and of biological stressors (chapter 6), while providing a basis for the discussion of uncertainty and risk integration (chapters 8 and 9).

The particular goal of this chapter is to explore the principles, databases, and models (tools) by which to characterize the diverse effects dealt with in ecological risk assessment. We propose to summarize and quantify, in so far as that may be possible, information on the following seven elements of effects characterization:

- Theoretical and conceptual foundations;
- Common methods and approaches available;
- Stressor(s) and assessment endpoint relationships, including evaluation of available ecological effects data;
- Patterns of responses to one or more stressors;
- Tools for ecological response analysis;
- Relationships of measurement endpoints to assessment endpoints, including extrapolations and methods for considering causality; and
- Development of approaches to stressor-response profiles.

As outlined in the *Framework Report* (U.S. EPA, 1992a), the elements of effects characterization must be thought of as a set of measurements or computations that are interactive with all the other components of risk assessment, including significance determination, problem formulation, and, ultimately, risk characterization. Thus, the tools and approaches discussed in other chapters are relevant here, but need not be repeated. Especially relevant is the chapter on biological stressors (chapter 6), a particular class of effects that will be noted only occasionally in this chapter.

Much of the rationale and methodology for ecological risk assessment has come from health risk assessment methods (NRC/NAS, 1993) where "effects characterization" is not a major issue in its own right. Advanced approaches suitable for ecological systems have been developed recently, partly through improvements in data and predictive tools (Suter, 1993), but also because of the acceptance of fundamental differences between human and ecological systems. Legislation such as the National

Environmental Policy Act (NEPA) of 1970, the Endangered Species Act of 1973, the National Forest Management Act of 1976, and the National Acid Precipitation Assessment Program (NAPAP) from 1980 to 1990 have all directed that the potential for significant negative effects from human intervention in natural resource systems be considered (Loucks, 1992). These directives have led to the development of many new databases and assessment methods. In this discussion, consideration also is given to the greatly improved theoretical and empirical foundations now available for considering indirect as well as direct effects over long periods of time (Sheehan, 1984a, b, c; Shriner et al., 1990; Suter, 1993; NRC/NAS, 1993). Thus, the sections in this chapter are organized on conceptual foundations that serve as the basis for considering the approaches available, the patterns of responses to one or more stressors evident in data, and relationships of observed effects to measurement and assessment endpoints.

2. CONCEPTS AND TERMINOLOGY

Conceptual issues and definitions of terms must be considered together in characterizing the ecological effects of chemical and physical stressors. Terms relating to the framework are considered in chapter 1 (see also U.S. EPA, 1992a), but we need to be more explicit about direct versus indirect effects, scale considerations in space and time, and thresholds in discussions of characterizing ecological effects.

2.1. General Considerations

Consensus is needed on the of meaning of many new terms and concepts that address evaluation of ecological effects induced both directly and indirectly by exposures to chemical and physical stressors. These include problems in quantitatively distinguishing effects against a background of natural biological variability; the influences of spatial and temporal scales on assessments at various levels of biological organization; and specific theoretical, conceptual, and measurement considerations involved in characterizing stressor-effect relationships for individuals, populations, ecosystems, and regions.

2.1.1. *Direct and Indirect Effects*

An effect is a change in the state or dynamics of an organism, population, or ecological system resulting from exposure to a chemical, physical, or biological stressor. The effects of chemical and physical stressor exposures on biota and ecological systems may be either direct or indirect. Direct effects are those that can be related causally to exposure of the stressor. For chemical exposures, direct effects are those mediated by the molecular interaction of the chemical and the reception site. The direct effect of physical stressors on individuals and populations are mediated by changes induced in physiology and behavior that reduce survival or reproductive success. Direct effects are illustrated by many of the outcomes commonly assessed in ecotoxicological studies. These include, for example, the mortality of individuals and reductions in population abundance associated with exposure to a chemical at a chronic-effects concentration and the reductions in reproductive success and early age-class recruitment of populations of species such as the California condor, which requires large undisturbed habitats but has historically been exposed to physically disturbed habitats.

In contrast, indirect effects are not mediated directly by the interaction of the stressor and the receptor but by biological interactions with the directly exposed individuals or populations. Some of the relationships between direct and indirect effects are shown in figure 5-1.

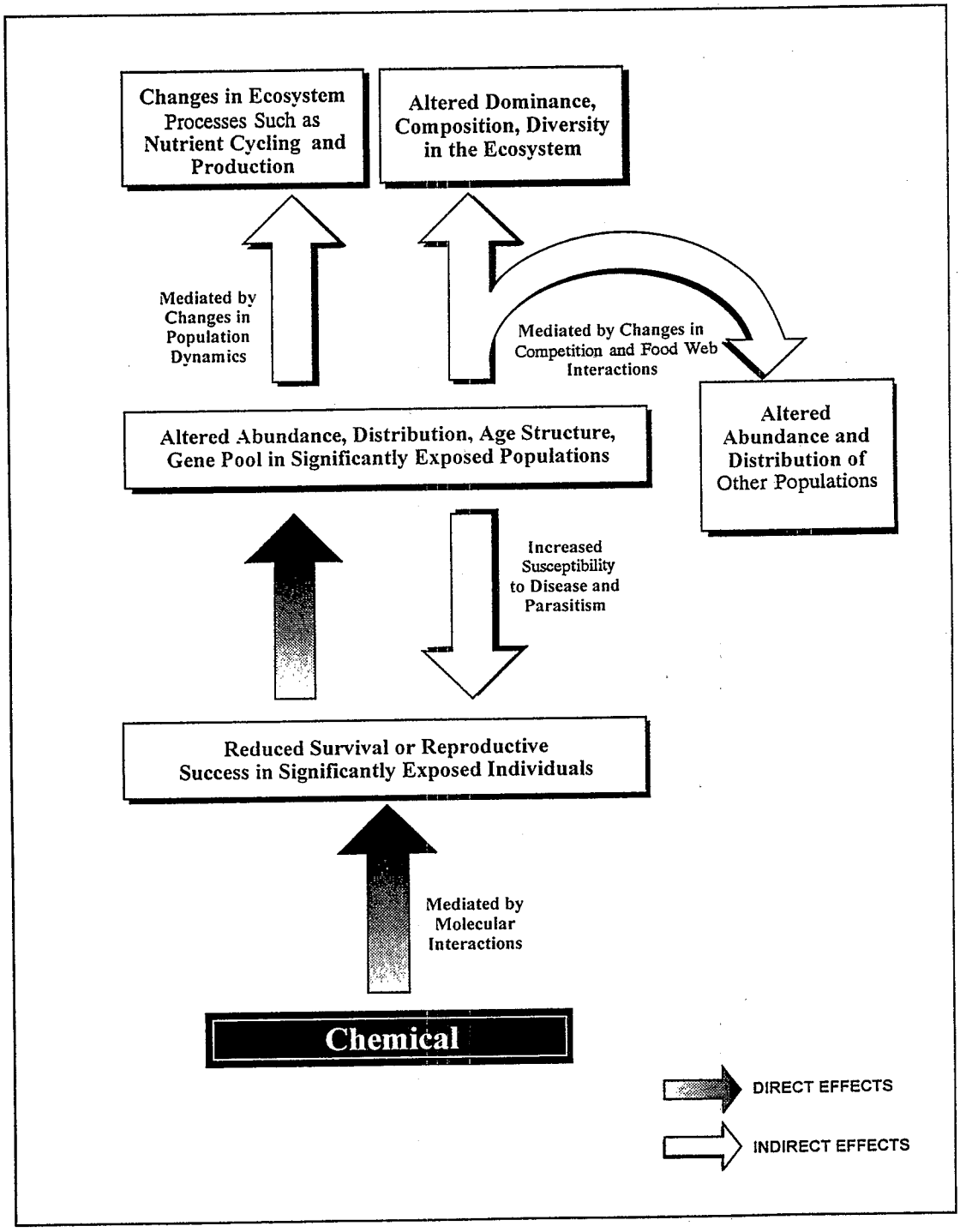

Figure 5-1. A schematic of selected relationships between direct and indirect effects of chemical exposures.

An assessment of aerial insecticide spraying in the Canadian prairies (Sheehan et al., 1987, 1994a) provides an example of the evaluation of indirect effects. The assessment found that aerial spraying of insecticides may produce direct toxic effects on exposed aquatic macroinvertebrates in ponds without resulting in direct effects on avian species because of the low relative toxicity of these chemicals. Nonetheless, secondary indirect effects from wind-spread aerial insecticide applications may include reduced recruitment of ducklings if macroinvertebrate prey abundance (a key food resource) is reduced substantially by the insecticide during the post-hatch period (when ducklings are obligate feeders on these organisms).

Indirect effects may move along any of the pathways connecting the directly affected population with other populations in the ecosystem. Direct and indirect effects may occur within the same timeframe of the stressor exposure, but indirect effects tend to be long lasting and can persist well after the direct stressor effects have attenuated in the ecosystem. Thus, both direct and indirect effects can be ecologically significant and need to be considered in ecological risk assessments.

2.1.2. *Distinguishing Ecological Effects of Chemical and Physical Stressors*

Fundamental difficulties for quantitative examination of ecosystems arise from the inherent biological variability of natural systems related to cyclic, stochastic, or nonlinear dynamics of populations. Our ability to describe many biological phenomena also is limited by practical constraints of sampling time and area. We are at times unable to apply measurement strategies appropriate to resolving natural versus induced population trend patterns in either experimental or monitoring studies. One problem is in deciding whether an observed change in the state of a system is a deviation caused by the presence of a toxic chemical, physical disturbance, or biological perturbation, or is part of the natural fluctuation inherent in the ecosystem dynamics.

A number of biotic and abiotic factors govern the state of an ecosystem after exposure to a stressor, including substrate type, distance from colonizing sources, and genetic variability of resident populations. Also, a number of endpoints may be used to define an ecosystem's state. Indeed, data for any one endpoint alone will provide a limited (and possibly distorted) view of the dynamics of a system's response to a stressor. Thus, multivariate analysis of data is often necessary to document anything other than gross changes in system state.

Another important concept in distinguishing ecological effects is that ecosystems are not inherently stable or governed by equilibrium. Rather, ecosystems are dynamic and thus in a state of constant change. Therefore, it is important to define statistically the difference between reference and

exposed system states at any point in time, as well as differences in the dynamics of the compared systems through time.

The question of distinguishing ecological effects attributable to chemical, physical, or biological stressors from inherent variability is central to characterizing effects and estimating risks to ecological systems. Still, our understanding of the key ecological and statistical issues often has been inadequate. Hurlbert (1984) described some of the experimental design problems in ecological studies and presented data on the problems arising from misuse of inferential statistics. Adequate baseline measurements are not available in many situations, at least not measurements sufficient to allow definition of the types and sizes of inherent fluctuations in species and process rates. Notwithstanding the significance of this problem, an equally important issue for design of laboratory and field experiments involves the testability of hypotheses involving fluctuating systems.

2.1.3. *The Concept of Threshold*

The concept that there is an exposure threshold below which no measurable effects, or no biologically significant effects, occur in the individual is the basis for much of the regulation of toxic substances in the United States. Although the concept of threshold in ecotoxicology has been challenged, possibly because it appears permissive of some level of a chemical or physical stressor exposure, it is implicit in much ecological literature and explicit in toxicology and the regulation of chemicals (Woodwell, 1974; Cairns, 1977, 1992; Odum et al., 1979). To characterize the ecological effects associated with exposures to various stressors, scientists have to address the theoretical and practical problems of identifying thresholds. Indeed, there is no single threshold applicable to all stressors, species, and assessment endpoints. Rather, appropriate response curves, sometimes including thresholds, specific to the assessment objectives, stressor, and assessment endpoints of interest will need to be identified to support effects characterization.

A second issue is whether to select a threshold based on our ability to measure it (e.g., a statistically significant change in the individual or population) or based on the biological significance of the effect (e.g., a change that affects a population's potential to survive or function within the ecosystem). The discussion of significance in chapter 2 is relevant here as well. As noted above, distinguishing a change is a serious sampling and statistical challenge in fluctuating and dynamic systems. The task of distinguishing a biologically *significant* effect has the added complication of assigning a value to the assessment endpoint (see chapter 2). A small but statistically significant decline in a single microbial population might be considered insignificant socially, while a dramatic

decline in species richness and the abundance of all decomposers would be regarded as highly unfavorable, especially if such a decline affected the litter decomposition rates and nutrient cycling of the exposed ecosystem.

2.1.4. *Scale Issues Associated With the Level of Biological Organization*

The effects of chemical and physical stressors may be assessed in terms of measurements at a variety of temporal and spatial scales applicable to individuals, populations, ecosystems, or regions, or some combination of these levels of biological organization. Scale here is defined as the relative extent in area or over time. Although stressor effects on molecules, cells, tissues, and organs can be measured, their significance in ecological risk assessment may be modest, except for their use in the form of biomarkers that help to relate biochemical effects to whole organisms and populations. On a human time scale, a population of individuals (of a species) is the smallest persistent biological unit of interest. Ecosystems can be viewed, to some degree, as aggregates of populations functioning together; however, ecosystems also have unique structural features (e.g., spatial structure, diversity) and functions (e.g., nutrient cycling, food web relationships) that have to be considered differently over long time scales. Although in the context of risk assessment, regions are spatial aggregates of contiguous ecosystems, data on populations, over short time scales, are important at these large spatial scales. Regions also have both the properties of their constituent ecosystems and unique attributes due to the patterns of ecosystem interactions at a regional scale.

Although both measurement and assessment endpoints can be defined at various levels of biological organization, the endpoints are not equally important over the range of spatial and temporal scales (Suter, 1993), as is shown in figure 5-2. At short temporal scales (days to months) and small spatial scales (micrometers to meters), effects of chemicals can be assessed on microorganism populations. However, little can be predicted about the ultimate effects of chemical and physical stressors on long-lived organisms at a scale of ecosystems, most of which function primarily at large spatial and long temporal scales. Since ecosystem functions are carried out by individuals that make up populations within those ecosystems, the distinctions between levels of organization are not as real as they seem.

Effects on individual macroorganisms can be expressed over a wide range of spatial and temporal scales. Small animals and plants in both terrestrial and aquatic systems operate within a scale of a few meters and generally within a temporal scale of days to months. Large animals and plants (e.g., birds and trees) operate within spatial scales of meters to tens of kilometers and a temporal scale of

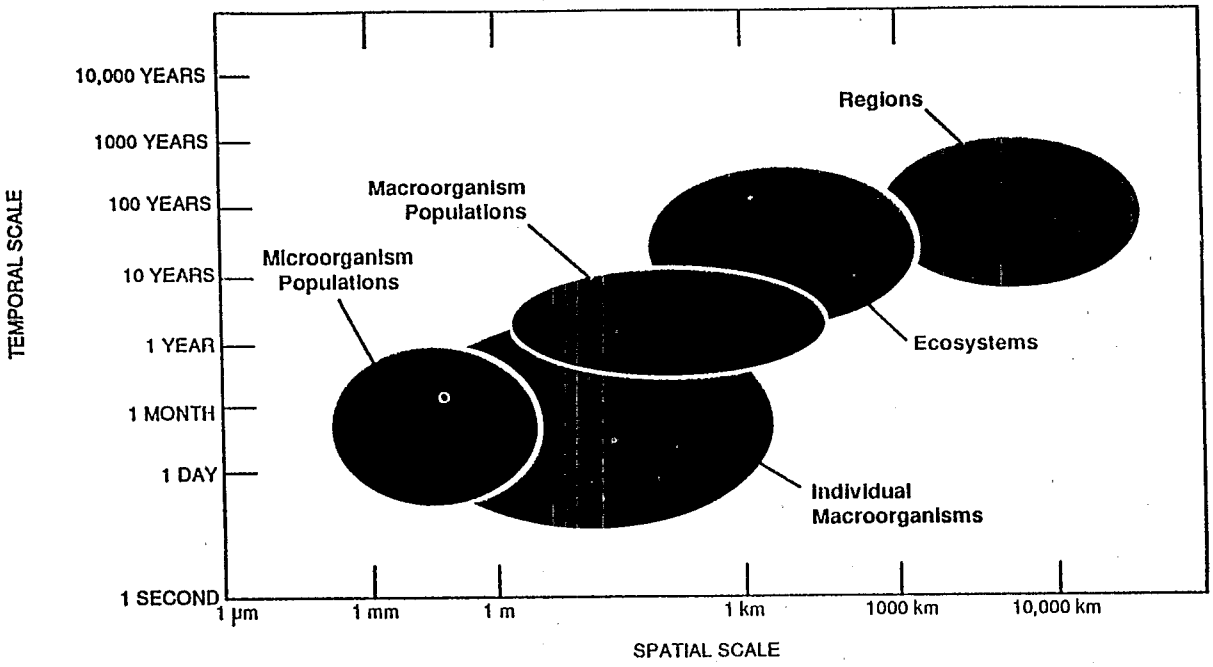

Figure 5-2. **Spatial and temporal scales within which individuals, populations, ecosystems, and regions respond to physical and chemical stressors, as shown by the overlapping ellipses (modified from Sheehan, 1994).**

months to tens of years or centuries. The effects of chemicals on populations of organisms usually can be assessed only within a temporal scale of months to years and a spatial scale of meters to kilometers. Direct chemical effects on one or more populations may, however, affect other populations in the exposed ecosystem. Such effects can occur indirectly through changes in habitat availability or through predator-prey or competitive interactions with other species. Because still other secondary or indirect effects may occur over longer time scales, and therefore may spread to larger areas, the range of spatial and temporal scales for evaluating chemical effects on ecosystems is broader than for the assessment of effects on populations alone (i.e., years to hundreds of years and meters to hundreds of kilometers). The spatial scale of regional responses to chemical exposure overlaps with but extends beyond that of individual exposed ecosystems (hundreds of kilometers to thousands of kilometers). Other aspects of spatial and temporal scales associated with ecological effects are discussed by Sheehan (1984a, b, c, 1991) and Suter (1993).

As noted in chapter 4, exposures to chemical and physical stressors also occur over a range of spatial and temporal scales. Clearly, the scales of exposure need to be related to the scales of biological organization in order to assess the risks of ecological effects. Single local applications of pesticides often occur on a spatial scale of meters in residential areas, to perhaps a kilometer in agricultural settings, over a temporal scale of hours to a few days. Chemical and petrochemical spill events can occur on a somewhat larger spatial scale, but the residual contamination from these spills may cause exposure on a temporal scale of days to years. In contrast to these local chemical releases, broad-scale applications of pesticides, such as aerial spraying to control grasshoppers in the prairie regions of the United States and Canada (Sheehan et al., 1987) and the spruce budworm in the forests of New Brunswick (Mitchell and Roberts, 1984) covered hundreds to thousands of kilometers in space and occurred intermittently over years to tens of years. At the extreme, widespread aerial transport of photochemical oxidants and acids has led to stresses in large geographical regions (thousands to tens of thousands of kilometers) as reported by Shriner et al. (1990). Cumulative exposures to airborne chemicals (and aqueous effluent discharges within watersheds) also take place over long periods (tens to hundreds of years). A more detailed discussion of the scale of chemical hazards can be found in Suter (1993) and Sheehan (1994).

2.2. Linking Stressors to Effects on Individuals and Populations

Most methods for characterizing the ecological effects of chemical and physical stressors, and most applications of these methods for assessing ecological risks, have focused on populations, while

recognizing that causal processes operate through individuals. Although a focus on the individual in ecological systems is analogous to the focus on the individual in human health assessments, the utility of individual-based assessments is more limited in ecotoxicology. The status of individuals is clearly of importance in the case of threatened and endangered species. In most other situations, however, it is the population that is the persistent biological unit and the focus of environmental protection.

The effects of stressors are often first manifested in individual organisms, or in an individual's ability to reproduce successfully. Thus, effects on populations are an integration of the stressor effects on the performance of individuals. The significance of the effect on one level of biological organization is typically assessed at the next higher level. For example, effects on individuals are biologically significant if they are lethal or if they reduce the reproductive success of the organisms in replacing themselves in the population. Also, effects on populations are biologically significant if they substantially reduce the natural abundance and/or alter the distribution of the population so as to reduce its functional role within ecosystems. Such changes in abundance, even when small, can have a profound effect on one or many populations initially, due to the characteristic nonlinear dynamics (chaos) evident in the interactions among some species (Logan and Allen, 1992).

In laboratory tests for characterizing the effects of stressors on macroorganisms, effects are expressed through the actions of individuals in a sampled population, even though the outcome often is viewed as a population effect (e.g., as an LC_{50}). Only large-scale laboratory tests on microorganisms are likely to be considered direct population-level tests. Therefore, a key issue in effects assessment is the extrapolation of test data for individuals to predict effects at the population (and ecosystem) level. The importance of field studies in characterizing the effects of stressors on higher levels of biological organization is reinforced by our present inability and lack of resources for conducting population-scale studies in controlled facilities.

2.3. Linking Stressors to Effects on Community and Ecosystem Structure

The structure of communities and ecosystems is defined by the abundance and biomass of populations and their spatial, taxonomic, and trophic organization. The responses of the populations in an ecosystem to an external stressor will be seen as a change in ecosystem structure. These changes may include reduction in population size or eventual extinction, reduction in species richness, changes in species dominance and diversity, and changes in spatial structure. Although such changes in ecosystem structure may reflect exposure of some populations to a particular stressor, they do not necessarily forecast the internal mechanisms by which the change in ecosystem structure is expressed.

The presence of nonlinear dynamics among the populations, as noted above, leads to large uncertainties as to outcomes at the community or ecosystem level (Yodzis, 1988).

The induced changes in structural properties have been the changes of primary interest in characterizing effects on ecosystems, rather than changes in specific functions of the ecosystem itself, although, obviously, these are ecologically important as well. To assess change, an appropriate set of structural references is essential. Baseline data on an ecosystem's state prior to stressor exposure can provide an ideal control for comparisons, but such data are seldom available. A time series of changes in structural indices after introduction of a stressor also would reflect effects on ecosystem structure. Reductions in numbers, biomass, and taxonomic and trophic diversity indicate a short-term disruption in transient or stability conditions. Conversely, increases in these indices with time would suggest at least partial recovery of the system. If baseline data are unavailable (as is frequently the case), changes still can be monitored over long periods of time. Although no set of values would be available against which to compare, the approach can be used to monitor changes in conjunction with stressor abatement.

Some ecologists have argued that structural indices best meet statistical criteria for the monitoring of functional responses of ecosystems to toxic substances. These indices usually do not have the daily periodicity of primary productivity or the short-term variability associated with respiration. Ecosystem structure, however, is only loosely related to ecosystem function. Ecosystem structure (e.g., diversity and dominance patterns) may change significantly under stress with little evidence of accompanying disruption of functions (e.g., productivity), or, conversely, functions such as mineral cycling can be altered without significant near-term changes in species composition and diversity (Matthews et al., 1982). The lack of a predictable relationship between structural and functional responses to stress suggests the need for a balanced approach, considering both structure and function, in assessing stressor effects at the ecosystem level.

Structural characteristics and the numerical measures dependent on them provide a range of information differing widely in ecological value. For instance, it may be more informative to know about changes in the taxonomic or trophic composition of an exposed ecosystem than merely the changes in biomass or the general abundance of organisms. Winner and Bewley (1978) and others have shown that a "diversity index" is insensitive to air pollution effects on species except under the most severe stress, while relative cover and the number of species per quadrant are reasonably sensitive. Also, structural indices do not necessarily follow similar patterns of change under different types of induced stress. Hellawell (1977) described several possible alterations in an ecosystem that

would be reflected in biomass only; in biomass and relative dominance; or in biomass, dominance, and composition. Because of the differences in (1) the value of information provided by these indices, (2) the ease with which they are measured or calculated, and (3) the sensitivity of their response to stress, the selection of ratios or other measures of ecosystem structure to assess ecological risks should be conducted thoughtfully and be based on the objectives of the assessment and the sensitivity, general applicability, ease of measurement, and ecological meaningfulness of the measure.

The effects of external stressors and their implications for biological diversity of ecosystems present a special problem in regard to air pollutants (Barker and Tingey, 1992; Armentano and Bennett, 1992) as well as toxic chemicals in water. One may appreciate the particular problems concerning applicability of data, interpretive aids, and indices by looking more closely at two expressions of biological diversity. Figure 5-3 shows data on the abundances of all the species of Staphylinids observed in soil cores from oak-hickory forests where acid precipitation appears to have induced lower pH values in the surface mineral horizons at two eastern sites (pH 3.9 in Ohio vs. 4.5 in Illinois) on historically identical soil and substrates. The Shannon-Weiner index of diversity (H′), a common "index," computes to values of 0.84, 0.76, and 1.06 for the three sites, but provides no means of evaluating the significance of the numerical differences. On the other hand, inspection of the three distributions of species abundance over species abundance rank order (figure 5-3), shows more clearly that the two eastern sites are different from Illinois in both abundances and numbers of species, with a pattern that is itself informative compared with the single index. Thus, in seeking to understand stressor effects on communities and ecosystems, one should not let original data be disregarded in favor of synthetic measures of effects, even when the synthetic measurement also is critically important. The ultimate problem, "missing species," is difficult to quantify, however, and means for observing their effects on ecosystem processes are only now being developed.

2.4. Linking Stressors to Effects on Ecosystem Function

During the 1970s and 1980s, a series of papers began to show that ecosystem regulation, like cellular and organismal regulation, is achieved often through the activity of small but critical constituents in ecosystems functioning hierarchically (Shugart et al., 1976; Loucks, 1985; O'Neill et al., 1986; Vanni et al., 1992). External stressors that subtly alter these regulatory agents and rates magnify the linkage and the effects of the stressor on system function (Sheehan, 1984c). The series of case studies reviewed in the report *Issues in Risk Assessment* (NRC/NAS, 1993) shows the prospect

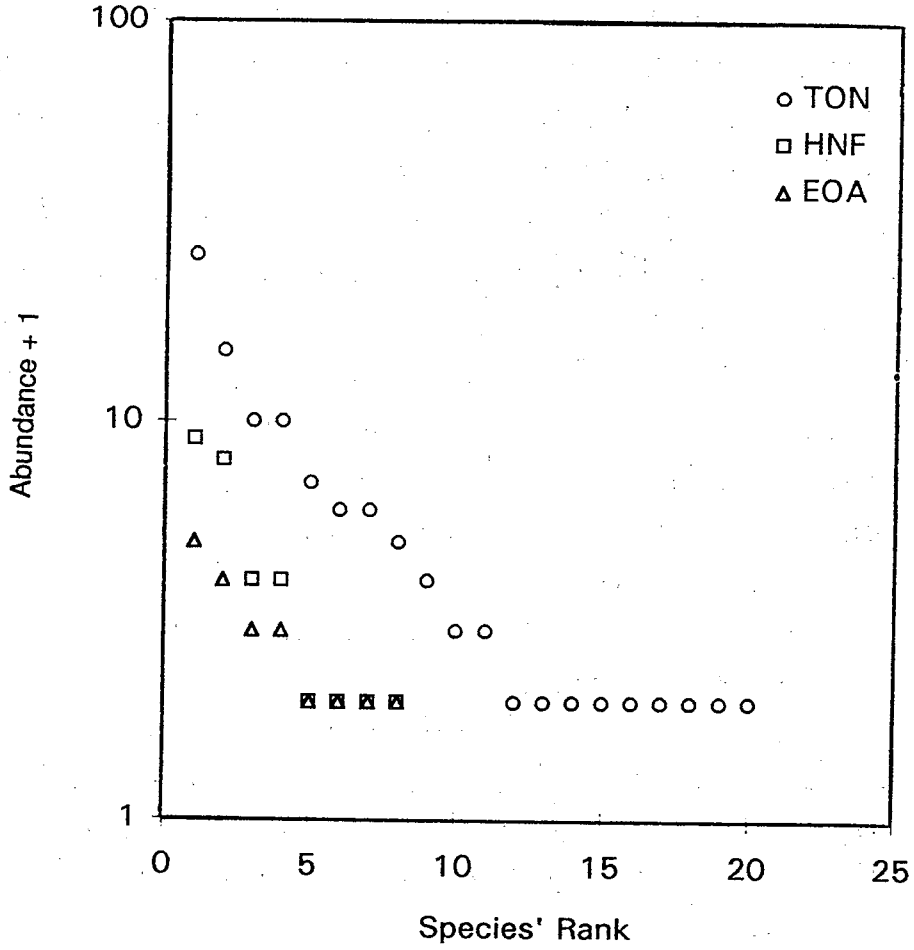

Figure 5-3. **A plot of the numerical abundances of soil Staphylinid species over abundance rank order for populations in soil cores at three Ohio Valley sites, for comparison with the Shannon-Weiner species diversity index (H′): Touch-of-Nature (TON), Illinois, 1.06; Hoosier National Forest (HNF), Indiana, 0.76; and Edge-of-Appalachia (EOA), Ohio, 0.84 (Sugg, 1994).**

of using our preliminary understanding of linkages within ecosystems to assess risk attributable to the alteration of feedback rates in ecosystems.

The complexity and incompleteness, to date, in theoretical and empirical foundations for predicting stressor linkages is especially evident in the population- and ecosystem-level effects literature reviewed by Suter (1993). The species providing specific ecosystem functions, as well as the sensitivity of these species to stressors (e.g., soil fauna functioning in nutrient and organic matter cycling but stressed by agricultural chemicals), are largely unknown and are only now beginning to be investigated quantitatively. A kind of experimental "sensitivity" analysis may be needed in large-scale field studies to improve our understanding of risks from stressor effects on ecosystem processes and function.

2.5. Stressor Effects on Ecological Pulsing and Trajectory Patterns

Ecological systems are understood now as usually having some yearly, shorter, or longer-term pattern of fluctuation or trajectory (as in succession). While the yearly pulses are often satisfactorily understood in aquatic ecosystems, they are not the same each year and the significance of natural deviations from the average pattern is not understood. Each yearly or longer pulse induces a transient biological response that over short periods or in local areas may be viewed as a trajectory within which resilience or perturbations operate to restart the system. Averages over long observation periods, or over large areas, with many local transients can be viewed as a regional ecosystem with properties that approach stability. Such averages have certain characteristics of equilibrium conditions, but in the presence of external or anthropogenic stressors, the processes controlling fine-scale perturbations and short-term transients can be changed. The risk of destabilizing the entire system can be high, as is evidenced by the profound changes in the Georges Bank fishery (Fogarty et al., 1993). Excursions outside the range of resilience for the system can be induced, creating a potentially degraded and irreversible new system. The introduction of undesirable alternate quasi-stable states, and possibly chaotic responses, is probable under some circumstances. Our understanding of these risks, however, is still too incomplete to prescribe a standard methodology.

Pulsing and trajectory departure characteristics need to be considered in terms of a probability distribution of some form. Since the transition to a new quasi-equilibrium may run for several decades, the transition period (i.e., during which the ecosystem is not in an historical equilibrium or trajectory, a period that the risk manager may be confronting now) may be viewed as a time of destabilization. Under these circumstances, current ecological science can provide little predictive

capability. Many terrestrial wetland and aquatic resources are already in a transient response to environmental stresses and, therefore, research on theory and models for characterizing risks to nonequilibrium systems should be given a high priority.

Before trying to detail the role of anthropogenic stressors in these systems, however, the full range of natural stressors and perturbations that induce fluctuations or long-term natural trends must be quantifiable. Although this is beginning to be feasible for freshwater environments, it may be some way off for many terrestrial systems, considering that interactions between drought, insect, and disease outbreaks for the various age classes of forest are not well understood even for unstressed systems. The effects of stressors, while likely to change these natural interactions in some direction (Yodzis, 1988) may be beyond quantification for risk assessment purposes—at least at this time.

3. EFFECTS CHARACTERIZATION

Effects characterization is the qualitative and quantitative description of the relationship between the stressor and response (effects) in the exposed individuals, populations, or ecosystems. This section address endpoints, types of effects data, and evaluation methods.

3.1. Endpoint Context

Endpoints are formal expressions of the environmental values to be protected (Suter, 1989), and these endpoints must be explicitly defined in assessing ecological effects. The response of the ecosystem to the stressor is assessed in terms of these endpoints. A detectable change in an endpoint that indicates an alteration in the ecosystem is an "effect"; however, the biological and social significance of the change (i.e., whether it is an "adverse" effect) can only be judged with respect to the viability of the exposed individuals, or success of the populations, or the functioning of the ecosystem and the value society attaches to the amenities changed. Establishing endpoints requires identifying valued attributes of the ecosystem that are considered to be at risk and then defining these attributes in operational terms (Suter, 1993). Since general regulatory statutes broadly define the environmental attributes to be protected, more specific operational definitions of endpoints are essential to a useful risk assessment. Without clear definitions, endpoints provide no clear direction for testing and modeling. For example, a general goal such as "ecosystem health" is inadequate for risk assessment.

Suter (1993) has identified five criteria that any endpoint should satisfy:

- Societal relevance,
- Biological relevance,
- Unambiguous definition,
- Accessibility to prediction and measurement, and
- Susceptibility to hazardous agents.

Societal relevance indicates that the ecosystem attribute should be understood and valued by the public and by decision makers. Because we cannot practically study all species or ecosystem properties in exposed ecosystems, focusing on endpoints with societal relevance makes good sense. This approach also best serves the managers of environmental quality and biological resources, who require a similar focus to be successful.

As stated previously, the biological significance of effects in an endpoint at one level of biological organization often are measured by subsequent impacts propagated at higher levels. For

example, a change in the survival or reproductive success of individual organisms is significant if it affects the abundance or distribution of the population, and a decrease in population abundance is significant if it affects the structural or functional relationships in the ecosystem. Because of functional redundancy and other compensatory relationships in ecosystems, not all changes to populations necessarily affect ecosystem function. Therefore, endpoints may be identified that have societal relevance but little biological significance (e.g., the loss of a small number of redwood trees). Conversely, endpoints with biological relevance may not be perceived as important to society (e.g., the loss of a keystone species such as the starfish in intertidal habitats). Clearly, endpoints with both biological and societal significance should receive the utmost consideration for characterizing ecological effects. An expanded discussion of the ecological significance of effects of stressors is presented in chapter 2.

Finally, if the response of the endpoint to a stressor cannot be measured or estimated from the measurement of related responses, or if the endpoint is insensitive to the stressor, then the endpoint cannot be assessed. The best endpoints are those that are sensitive to the stressor and for which there are well-developed test models and field-measurement techniques.

There are two types of endpoints: assessment endpoints and measurement endpoints. Assessment endpoints are the characteristics of the ecosystem to be protected. They are often defined at relatively large scales and therefore are not easily measured directly (e.g., tree production in forests or fish production in the Great Lakes). Measurement endpoints are the laboratory and field data that can be extrapolated to characterize the assessment endpoints that cannot be measured directly (e.g., LC_{50} data for a fish species or growth rate data for a specific tree species under well-described greenhouse or field-exposure conditions). Examples of assessment and measurement endpoints are presented in table 5-1.

Characterizing the effects of stressors requires the identification of explicit assessment and measurement endpoints and methods to extrapolate from measurement to assessment endpoints. To facilitate the comparison of effects from different stressors, or to integrate the effects of multiple stressors, comparable endpoints must be used. For example, the effects of exposures to physicochemical and biological stressors can all be assessed in terms of changes in population recruitment or abundance, but cannot be assessed in terms of chemical LC_{50}s since there is no equivalent measure for physical or biological stressors. The selection of endpoints is discussed in detail in chapter 3.

Table 5-1. Examples of Possible Assessment and Measurement Endpoints for Evaluation of the Effects of Insecticide Spraying for Spruce Budworm Control (Sheehan, 1994)

Problem	Assessment Endpoint	Measurement Endpoint
Possible nontarget effects of long-term application of insecticides to regional forests to control spruce budworm	Probability of >10% reduction in salmon populations in streams in the sprayed area	LC_{50} or NOAEL for salmon or related fish species
	Significant decrease in tree canopy bird populations	Dietary LD_{50} for Japanese quail egg hatch and fledgling
		Success in treated and reference areas
		Population numbers for selected bird species in treated and reference areas
	A 20% decrease in fruit production from bee- pollinated plants	LC_{50} for bees
		Abundance and diversity of natural bees
		Populations of selected bee species in treated and reference areas
		Fruit production (e.g., blueberries) in treated and reference areas
	Significant decrease in forest litter decomposition	Microbial respiration in soils from treated and reference areas
		Soil arthropod abundance in leaf litter in treated and reference areas

NOAEL = No Observed Adverse Effect Level
LC = lethal concentration
LD = lethal dose

3.2. Type of Effects Data and Study Methods

The types of responses that can be used to characterize the effects of stressors on individuals, populations, and ecosystems are reviewed in Moriarty (1983), Sheehan (1984a, b, c), U.S. EPA (1988a), and Suter (1993), and summarized in table 5-2. The projected change (i.e., either an increase or a decrease) in response associated with the magnitude of the stress regime is what interests the risk assessor.

Data with which to assess the effects of stressors come from three sources: controlled laboratory and field studies, studies in stressed and matched reference (unstressed) ecosystems, and studies associated with environmental accidents (e.g., oil spills). Methods for measuring the effects of chemicals on aquatic organisms (Persoone et al., 1984; Waldichuk, 1985), terrestrial animals (Dobson, 1985), and aquatic and terrestrial plants (Calamari et al., 1985; Kozlowski, 1985) were reviewed in the 1980s. A variety of multispecies tests also were reviewed during this same period (Hammons, 1981; Cairns, 1985, 1986).

The preponderance of effects data for chemicals is from toxicity tests conducted either with the chemicals of interest added to water, sediment, or food, or from tests of direct exposures to contaminated water, sediments, or soils. Standard U.S. EPA methods exist for measuring the acute and chronic toxicity of effluents and receiving waters to freshwater, estuarine, and marine organisms (U.S. EPA, 1991a, b, c). The American Society for Testing and Materials (ASTM) provides guides and establishes practices for measuring the response of algae, invertebrates, fish, and amphibians exposed to chemicals in water and for conducting sediment toxicity tests with freshwater and marine invertebrates (ASTM, 1994). ASTM also establishes standard practices for conducting toxicity tests with rodent and avian species (ASTM, 1994). Additional guidance on conducting aquatic multispecies microcosm and mesocosm toxicity tests is provided in SETAC (1992a, b) and Hill et al. (1994). Models used to interpret toxicity test data are discussed in section 5 of this chapter.

The second largest set of effects data is gathered from field studies in which contaminated and reference sites are compared. Change in the exposed system is measured with respect to a baseline or some reference (e.g., the untreated control in a toxicity test, or an uncontaminated but similar stream, lake, forest, or watershed). That an effect can only be demonstrated by comparison with an appropriate reference is one of the principles of effects assessment (Green, 1979).

Several types of measurements may be collected to assess a single assessment endpoint. For example, to assess the effects of a chemical on the reproductive success of a species, one can measure the time to sexual maturity, the number of young per reproductive event, the number of reproductive

Table 5-2. Types of Responses to Stressors Characterized in Ecological Risk Assessment (adapted from Sheehan, 1991)

Response	Measurement	Comments
Impacts on Individual Organisms		
Reduced growth	Rate of change in size or mass of the organism	Growth is frequently related to development and the probability of survival
Altered development	Ability to develop to a mature adult stage or time to reach sexual maturity	Delays in reaching sexual maturity often translated into reduced fecundity
Reduced reproductive success	Time to first production of offspring, number of offspring per reproductive event, number of reproductive events per lifetime	Reduced reproductive success may mean a lower rate of recruitment to the population
Shortened life span	Length of life span as compared to normal length	Shortened life span reduces the individual's reproductive contribution to the population
Impacts on Populations		
Reduced abundance	Number of individuals or biomass of population versus chemical exposure	Must be compared to abundance of reference population in uncontaminated area or control population in enclosure study
Altered distribution	Presence/absence or commonness/rareness of a population versus chemical exposure	Analysis must account for life history and other ecological and environmental factors structuring population distribution
Changed age structure	Frequency distribution of age or size class versus chemical exposure	Can be used to assess recruitment success or recovery
Altered gene pool	Electrophoretic analysis of genotypic frequencies versus chemical exposure	Opportunistic species more likely than specialized species to develop resistant populations

Table 5-2. (Continued)

Response	Measurement	Comments
Impacts on Ecosystem Structure and Dynamics		
Population extinction	Absence of a population known to have existed prior to chemical exposure	Loss of "key" commercial or ecological populations is easiest to interpret
Changed community composition	Species list, indicator species, or indicator assemblages versus chemical exposure	Best to know what taxa are absent as well as what taxa are present; trophic organization may provide insights into effect on feeding relationships
Dominance switches	Relation of abundance versus chemical exposure	Abundance under chemical stress often depends on the opportunistic life histories of species
Changed diversity	Species-richness or species-abundance plots versus chemical exposure	Numerical indices are applicable only to gross levels of chemical pollution because conflicting data from richness and evenness components and insensitivity to moderate levels of pollution; species richness may be a more consistent measure of the effects of chemical exposure
Changed similarity	Coefficient of similarity, quotient of similarity, or percentage similarity versus chemical exposure	Measurements are more consistent and sensitive than diversity indices in assessing chemical effects on community composition
Reduced abundance/biomass	Total number or biomass of individuals in community versus chemical exposure	Measurements do not provide much information on the ecological character of the system, but these are the least expensive variables to measure

Table 5-2. (Continued)

Response	Measurement	Comments
Altered spatial structure	Vertical and horizontal patterns versus chemical exposure	Only of value in assessing pollution effects on nonmobile communities, such as forests that exhibit a distinct spatial structure
Impacts on Ecosystem Functions		
Reduced organic decomposition	Decomposition rate of plant litter or reference organic substrates versus chemical exposure	Effects may not be obvious until some time after exposure
Reduced nutrient conservation	Net loss of essential elements in mass balance studies versus chemical exposure; nutrient spiraling length in stream studies	Of most obvious importance in terrestrial ecosystems
Reduced primary productivity	Carbon-14 assimilation rate or other methods of measuring plant growth versus chemical exposure	Long-term reductions in primary productivity are the most obvious index of a functionally stressed ecosystem
Reduced ecosystem production	Net gain in ecosystem production versus chemical exposure (e.g., net oxygen evolution in an aquatic system); Odum's index of power may be appropriate	Integrates chemical effects on primary productivity
Altered food web and functional regulation	Changes in predator-prey or consumer-consumed interactions versus chemical exposure	The loss of a predator indirectly through the elimination of prey and the dramatic increase in a population after the removal of a predator or competitor are examples of chemical alternation of functional regulation

events per time interval (or lifetime), or the viability of the young produced. Although data on any one of these measurement endpoints may indicate a reduction in the reproductive success of the organism as a result of exposure to the chemical, concordance among data for several measurement endpoints is important for reducing uncertainty in characterizing the stress regime-response relationship.

3.2.1. *Individuals*

In ecological risk assessments, effects are measured primarily on individuals and populations. Biochemical responses (biomarkers) of fish and wildlife to chemical exposures can be used to verify or quantify exposures, at least on a relative scale, in individual animals across multiple species (McCarthy and Shugart, 1990; Theodorakis et al., 1992; Brewer and Hummell, 1993). At present, available data are insufficient to establish a relationship between biomarker levels and effects on the performance of individuals. Cell and tissue pathology have been used to demonstrate the incidence of chemically induced effects on exposed individuals. For example, the feasibility of using tumor incidence in fish as an indication of significant exposure to carcinogens in contaminated water has been examined (Sindermann et al., 1980). However, the attributes that reflect individual performance, and that are the most easily related to population fitness, are growth, survival to normal life span (vs. early mortality), and fecundity expressed in reproductive success (see table 5-2). A review of the effects of chemicals on individual performance was provided by Sheehan (1984a), and tests to evaluate the toxicity of chemicals to individual organisms were recently reviewed by Suter (1993).

Growth is the net result of many essential processes, such as consumption and respiration. As a summation of many factors, growth represents a useful integration of indices of physiological status that is applicable to multicellular organisms that have not yet reached their maximum biomass. For example, cell division rate in unicellular organisms is a useful measure of both growth and reproduction. A reduced rate of growth has been correlated with a reduced rate of survival in young birds (Street, 1978), as well as the inability to reach sexual maturity and a reduced reproductive success in a number of species (Buikema et al., 1980; Krapu, 1974, 1979; Donaldson and Scherer, 1983). Growth of individuals is measured in toxicity tests for larval fish (U.S. EPA, 1988b, 1991b, c) and aquatic vascular plants (U.S. EPA, 1982). Although growth is also an important index of performance for terrestrial vertebrates, standard testing procedures emphasize mortality and reproductive effects over growth and developmental responses.

Reproduction is the single most important function in the life cycle of an organism. Successful reproduction is essential to population recruitment and the continuation of the species. Therefore, the real test of long-term impacts from sublethal stress on exposed individuals is whether the individual is capable of reproducing successfully. As a means of ensuring perpetuation, certain species have evolved a strategy of shunting a larger than normal proportion of available energy into reproduction under stressful environmental conditions (Bayne, 1975).

The importance of reproductive damage to species survival has stimulated much current research. An evaluation of methods to assess the effects of chemicals on the reproduction function of a number of wild mammalian and nonmammalian taxa recently was prepared (Vouk and Sheehan, 1983) and should be of value in improving the assessment of reproductive damage. Reproductive failure for sexual species can occur during a number of processes: courtship, development of gametes, fertilization, embryo development, hatching (birth), and early growth. Since the reproductive process encompasses all life stages, its successful completion is a basic individual goal. The inability of an organism to successfully complete any one stage of the reproductive process would indicate a reduced reproductive fitness of the population.

As an endpoint, mortality (i.e., shortened life span) can be quantified readily in laboratory studies and in situations of gross field exposures. The value of recognizing what concentration/dose of toxic substances, or magnitude of physical stress regime, can cause a lethal response in an individual organism is obvious because shortening the normal life span of individuals (particularly a large number of individuals) can affect population abundance and reproduction potential. Mortality is an endpoint that can be measured for all life stages and species that can be tested. LC_{50} and LD_{50} data are the most commonly reported toxicity values (for populations). While they provide a substantial foundation of effects information for ecotoxicologists, these effects data are based on short-term laboratory tests with a limited number of species and chemicals. Considerable uncertainty is associated with extrapolations from toxic effects to "no effects" concentrations, from surrogate to native species, and from laboratory to field conditions. These extrapolation issues are discussed further in section 6 of this chapter.

Chronic toxicity tests for aquatic organisms include full life-cycle tests (egg to egg), partial life-cycle tests (adult to juvenile), early life-stage tests (egg to juvenile), and "short-term chronic" tests (eggs and larvae or just larvae). Several reviews of early life stage and other partial life-cycle tests have suggested that both provide similar estimates of "safe" concentrations if growth or reproductive success are assessed along with mortality (McKim, 1977; Macek and Sleight, 1977; Woltering, 1984).

The standard endpoint of chronic tests on fish has been the maximum acceptable toxicant concentration (MATC), more recently termed the chronic value by EPA (U.S. EPA, 1991b). The MATC is derived by hypothesis testing and has been criticized for not providing a consistent level of protection (Stephan and Rogers, 1985; Suter, 1993).

Full life-cycle and partial life-cycle (sensitive life stage) test data are available for few wildlife species or surrogates. For terrestrial mammals, data are largely limited to that generated for domestic mice and rats as a result of carcinogenicity bioassays. For birds, data exist for embryo toxicity (Hoffman and Albers, 1984) and reproductive toxicity (U.S. EPA, 1982; ASTM, 1994), but for very few species. Chronic tests to establish chemical dose-response relationships for birds are largely limited to ducks and quail. Almost no dose-response data exist for raptors and piscivorous birds that are the focus of many current food web risk assessments. The threshold for significant effects on avian reproduction is calculated using hypothesis testing statistics in a manner similar to the MATC in fish.

3.2.2. *Populations*

The population is the persistent biological unit and is the focus of most ecological risk assessments. Population endpoints include changes in abundance, distribution, recruitment, age structure (size distribution), and genetic composition. The population endpoints and methods for their measurement are shown in table 5-2. The interpretation of changes in population endpoints using life tables and other population models is discussed in section 5.

The premature death and reduced reproductive success of individuals are ultimately reflected in lower recruitment and abundance and altered distribution of the exposed populations. Abundance is the most commonly assessed population endpoint; however, it is not the easiest to interpret. Abundance is a dynamic property that may change dramatically during annual cycles or time cycles of other duration. Most point estimates of abundance provide only a "snapshot" of population size that can be compared with a prestress baseline or reference abundance level. Characterization of abundance changes attributable to stress regimes also must take into account other factors that influence abundance but are not related to the stressor(s) of interest. These may include immigration rate, emigration rate, predation rate, harvesting rate (hunting or fishing mortality), and food availability.

Population abundance estimates are generally based on field counts of population numbers. Because only in rare cases can all members of a population be counted, accuracy and precision of

abundance estimates depend on the sampling design used to collect these data. For organisms with multiple life stages, sampling all of the life stages is essential to estimating population abundance accurately.

Recruitment success is a clear indicator of the ability of the population to replace itself. Recruitment is a particularly important endpoint for assessing stressor effects on birds and mammals, where the number of young produced per female is small in comparison with other species. The number of juveniles reaching sexual maturity per parent is a measure of recruitment in avian and mammalian populations. Again, recruitment success in stressed populations is generally assessed with field studies of the number of individuals reaching maturity. An analysis of age/size distribution in macroinvertebrate and fish populations provides data on the size of the juvenile population approaching sexual maturity.

Stressor alterations of the population gene pool have been reported but are more difficult to quantify than other population endpoints. In addition, the significance of changes in genetic composition are difficult to judge within a short term.

3.2.3. *Community and Ecosystem Structure*

Several reviews have documented the principal features of ecosystem structure required in risk assessment (Smith, 1981; Sheehan, 1984b; Waide, 1988; Bartell et al., 1992). Others (e.g., Kitchell, 1992) have described these subjects in terms of community structure, including such concepts as top-down control and "cascading." Although communities and ecosystems are fundamentally rather different concepts, multispecies population ecology and species-sensitive ecosystem function research have lessened the need for explicit consideration of community-level effects. To this end, we will consider community and ecosystem structure as closely related concepts serving as elements in understanding effects of stressors on larger-scale systems.

Some of the endpoints used to assess effects on ecosystem structure are presented in table 5-2. These include loss of populations, alterations in diversity and dominance relationships, changes in abundances or biomass, alterations in spatial structure, and changes in stability or fluctuation patterns.

To understand what might be a significant, induced departure from "normal" ecosystem (or community) structure, one has to have considered the relationship between the means based on measurements (and variability) over short periods and their relationship to spatial and temporal scale (see Suter, 1993). Let us say it is sufficient for now to note that small-scale variability in responses (and in predictive capability) become embedded in larger-area and longer-term averaging. The goal

in risk assessment is to estimate how a stressor may have altered local or short-period responses (e.g., fruiting, fires, flooding), or the spatial distribution of these events, so as to bring about a change in the dynamics or mean endpoint expression in the larger-scale system. The remarkable range in the time scale of important responses is summarized in table 5-3.

Further components of ecosystem structure that are readily affected by anthropogenic stressors include the demographics, average life span, and pattern of senescence in long-lived species such as, for example, trout in large lakes and certain forest trees (Beeton, 1969; NRC/NAS/RSC, 1985; Smith, 1981; Pedersen and McCune, 1990). These effects can lead to changes in population abundances (or in community composition) that represent a significant alteration in ecosystem functioning. Examples include changes in water clarity (and therefore feeding) and nutrient cycling, in leaf production and leaf-litter quality, and in decomposition or other critical rates that affect mineral cycling and growth (Smith, 1981). Thus, because over long periods of time these anthropogenically induced changes in relative species dominance not only alter the structure of ecosystems but also their functioning, they are capable of inducing a new, potentially irreversible quasi-equilibrium in nutrient processing and related food chain productivity. The effects of chemical stressors on experimental ecosystems have been assessed with multispecies microcosm and mesocosm tests (reviewed by Cairns, 1985, 1986) and field studies of structure in exposed and reference systems (reviewed by Herricks and Cairns, 1982).

3.2.4. *Ecosystem Function and Regulation*

As previously suggested, the larger-scale and longer-term responses of ecosystem endpoints must—if causal mechanisms and predictive capability are to be recognizable— incorporate the altering of processes at the population and microbial level. Ultimately, it is our understanding of fine-scale responses to stressors that gives confidence to projections of effects on endpoints such as net primary production (NPP) or net ecosystem production (NEP) at large geographic scales. For example, ozone (O_3) effects on forests are probably important for their altering of certain productivity endpoints. Still, these effects can be estimated only partially from the known relationships between O_3 concentration and foliar carbohydrate loss or tree growth, by species and age class. Similarly, the productivity of rivers with a high sediment load is affected by the shading effects of sediment on photosynthetic rates, affecting each part of the food chain differently. The means for making computations of effects at the functional or system feedback (regulatory) level, and for aggregating them to the level of significant large-scale endpoints, are developing slowly. The approaches must be

Table 5-3. Simplified Conceptual Model of the Apparent Time Scales of Physiological and Ecological Processes Associated With Plant Community Responses to Chronic Air Pollution* (Armentano and Bennett, 1992)

Response Variable	Time-Scale Interval
Pollutant uptake	10^{-1} to 10^{3} minutes
Reduced photosynthesis; altered membrane permeability	10^{1} to 10^{3} minutes
Reduced labile carbohydrate pool	10^{0} to 10^{1} days
Reduced growth of root tips and new leaves	10^{1} to 10^{2} days
Decreased leaf area	10^{2} to 10^{25} days
Differences in species growth performance	10^{2} to 10^{25} days
Reduced community canopy cover	10^{2} to 10^{3} days
Reduced reproductive capacity	10^{2} to 10^{3} days
Shifts in interspecific competitive advantage	10^{2} to $10^{3.5}$ days
Alteration of community composition	$10^{2.5}$ to 10^{4} days
Change in species diversity	10^{3} to 10^{4} days
Change in community structure (physiognomy)	$10^{3.5}$ to $10^{4.5}$ days
Functional ecosystem changes (e.g., decline in nutrient cycling efficiency, net productivity)	$10^{3.5}$ to $10^{4.5}$ days

*Time scales are suggested for a hypothesized forest community exposed to chronic ozone levels comparable to much of the eastern United States of the 1980s. The time-scale intervals, which are not verified empirically and are not intended to be associated with a specific site, are suggested as the ranges within which response symptoms would be clearly detected given current capabilities in pollution effects research.

expanded greatly and tested, however, before guidelines for their use in risk assessment can become operational.

In addition to NPP and NEP, another important element of ecosystem functioning that is changed in many systems is formation of secondary metabolites such as phenol glycosides in the foliage and conductive tissues of many plants. Such compounds are critical in affording resistance to disease and insect attack; many stressors, however, such as O_3 and acid gases taken up in foliage, substantially alter the formation of secondary metabolites (Coleman et al., 1992). The outcomes of these interactions are not yet fully predictable, partly due to the very large number of insect and disease agents for which various resistance mechanisms exist. The likelihood of major change in ecosystem response patterns operates through such mechanisms, however. Changes in the availability of carbon (C) relative to nitrogen (N), either by the altering of net photosynthesis rates (e.g., through O_3 effects and carbon dioxide [CO_2] enrichment) or by increasing the availability of N in the nutrient cycle (rainfall inputs), can affect the C:N ratio of primary metabolites (Coleman et al., 1992). This affects production of the largely carbon-based secondary metabolites and, therefore, insect or disease defenses (especially following droughts).

Although abundant data are available on nutrient cycles in ecosystems, the risk assessor should be concerned primarily with the likelihood of changes being induced in nutrient cycling. Changes in the cycling of N, phosphorus (P), potassium (K), calcium (Ca), or magnesium (Mg) can result from apparently small shifts in ecosystem function. Some of these may be the result of selective effects by the stressors on key groups of species (e.g., on soil fauna or fungi), as well as on the physicochemical soil system. The potential for changes in the magnitude of the nutrient pools (or in their turnover times) needs to be given special attention, as was done in the work of Finn (1976) and Watson and Loucks (1979). These papers show highly characteristic patterns of N and P turnover in ecosystems. Many of the nutrient pools (e.g., dissolved inorganic P) have short turnover times (minutes) and are highly susceptible to stressors that then have a radical effect on the whole system. Risk assessment must focus on the likelihood of changes in these rates and turnover times (which are poorly known in terrestrial systems), rather than on the gross stocks and fluxes that have been the focus of most nutrient cycling studies.

Finally, an increasingly common measure of "system function" status for ecosystems is the Index of Biological Integrity (IBI), a measure of both food web connectedness and natural diversity (Karr et al., 1986; Miller et al., 1988). Major regional studies, however, have been required to ensure that IBI can be consistent enough in its use for regionwide application in streams. Its adaption to other

types of ecosystems is quite problematic at present. IBI is in the nature of a potentially useful, highly integrative endpoint, but in some of its current applications it falls short of having the features desired in an endpoint.

3.3. Evaluation Methods

Effects data can be evaluated on several levels to address specific questions. First, are the data of good quality? Second, are the data appropriate for the endpoint being assessed? Third, are the data representative of the individuals, populations, or ecosystems of interest? Fourth, are the data sufficient to meet statistical model requirements? Each of these data evaluation concerns is addressed in the following sections.

3.3.1. *Data Quality Assurance*

The goal of quality assurance in risk assessment is to build confidence in the data so that they will be maximally useful in the risk management process. Laboratory stressor-response studies and field effects characterization studies should incorporate sufficient quality assurance procedures to ensure data quality. These may include culture controls to evaluate test organism health; reference controls to evaluate organism response to an environment without the stressor of interest, but with other important environmental factors; and positive controls to evaluate response of the test organism to a single stressor (in comparison with historic test results with the same stressor). The quality of effects data can only be judged with respect to study controls. Therefore, the use and comprehensive characterization of laboratory and field control samples is key to the quality assurance of effects data. Quality assurance also includes identifying appropriate data quality objectives, providing proper documentation of studies, properly maintaining and calibrating instruments and equipment, avoiding bias in sampling and measurements, and providing accurate calculations and training personnel to meet job requirements.

Since both field studies and laboratory toxicity tests can be used to characterize effects, the work can be conducted in accordance with a comprehensive quality assurance and quality control plan. Such a plan for toxicity studies would describe in detail standard operating procedures for all toxicity tests, culture methods, equipment and instrument maintenance and use, glassware cleaning, sample collection, sample preservation, sample preparation, chain of custody, and chemical and data analyses (U.S. EPA, 1991d). Guidance on good laboratory practices for toxicity testing are provided by the U.S. Food and Drug Administration (U.S. FDA, 1987), EPA (Federal Insecticide, Fungicide and

Rodenticide Act [40 Code of Federal Regulations (CFR), Part 160], Toxic Substances Control Act [40 CFR, Part 792], and the Organization for Economic Cooperation and Development (OECD, 1981).

More limited guidance is available on the quality assurance requirements of biological sampling to support quantitative field studies of effects. The EPA Environmental Monitoring and Assessment Program provides field operations and safety manuals (e.g., U.S. EPA, 1991e).

3.3.2. *Matching Data and Measurement Endpoint to the Assessment Endpoint*

The more closely related the measurement endpoint is to the assessment endpoint, the less the uncertainty in extrapolation of the effects data in the risk assessment. In contrast, the more remote the effects data from the assessment endpoint of interest, the more uncertainty in the risk assessment. This is a problem evident in the greater use of acute exposure LC_{50} or LD_{50} data rather than EC_{10}, a Lowest Observed Effect Level (LOEL), or a No Observed Effect Level (NOEL) for predicting no effect concentrations for species of interest in risk assessment. Barnthouse et al. (1990) showed that for fish, effective concentrations for chronic responses can be predicted to no greater than two orders of magnitude precision in the absence of life-cycle or partial life-cycle data for the species of interest.

Similarly, the more closely related the test species to the species of interest, the less the uncertainty in the risk assessment. This is based on the assumption that a species can represent the taxon to which it belongs, but the uncertainty associated with prediction from species to species increases with increasing taxonomic distance. This generalization is supported for aquatic animals by the work of Suter et al. (1983), Sloof et al. (1986), and Mayer and Ellersieck (1986). The generalization that similarity in toxic response is related to taxonomic similarity will not likely hold if there is not a gradient of the traits controlling sensitivity to the stressor. This would be true either when all species have evolved a similar adaptation to a stressor, or when all species were equally unprepared to resist a stressor.

3.3.3. *Representative Data*

Although a substantial amount of data may be available for an area of concern, the data may have been collected for objectives other than risk assessment and thus its usefulness in a risk assessment may be limited. This is true of the qualitative biological survey data collected for environmental impact reports and one-time monitoring programs that lack the quantitative properties to be useful in an assessment of stress regime-response relationships.

Similarly, toxicity test data for which the exposure methods bear no relationship to potential environmental exposures are of limited usefulness in effects assessment. Included are data from tests based on chemical exposures via injection and other unnatural routes of uptake, which may be useful in determining the mechanism of toxicity, but not for determining a relevant dose-response relationship. Also important is matching the exposure pattern of toxicity tests to that expected in the environment. Many laboratory tests are currently conducted using a continuous exposure regime, whereas many chemical releases to the environment are periodic in nature. The use of data from laboratory or field studies on species unrelated to those at issue in the risk assessment is equally concerning, given the high degree of variability in stressor response among species, particularly species that are not taxonomically of functionally related, as previously described. A related problem are data collected from improper sampling designs. Biased sampling, or collection of samples of improper type or size, will provide unrepresentative estimates of population parameters.

3.3.4. *Methods to Evaluate Data Sufficiently for Statistical Models*

Two types of statistical models are used to characterize stress regime-response relationships: (1) hypothesis test models and (2) curve-fitting models. In the hypothesis test model, responses at the selected exposure concentrations are compared with control responses to test the null hypothesis that they are the same. In the curve-fitting model, a function is fitted to a set of points relating the measured effects to measurements of exposure based on the hypothesis that the slope of the relationship is not zero. These two models are used for different objectives: the hypothesis test model to assess differences in response between treatment and control, and the curve-fitting model to prescribe a level of effect within the range of exposure measured. Neither model is designed to make predictions outside of the range of test data. To extrapolate accurately to effects below the stressor levels tested requires a mechanistic model based on the mechanism of action rather than a statistical model. The advantages and disadvantages of the hypothesis test model and the curve-fitting model for analyzing toxicity test data have been discussed by a number of authors (Stephan and Rogers, 1985; Graney et al., 1989; Shaw et al., 1994).

The discriminatory power of both statistical models is dependent on the number of samples available for inclusion in the analysis and the experimental design. For the hypothesis test model, the power of the test to avoid false positives (i.e., identifying an effect when no effect exists) and false negatives (i.e., not identifying an effect when an effect exists) is related to the variability in test

(sample) data, the minimum level of discrimination desirable (e.g., 10 percent difference between treatment and control means), and the allowable level of false positives and negatives.

The sufficiency of data to support a test of selected power and confidence, or the evaluation of the power of a hypothesis test based on collected data, can be determined from various power law models (Green, 1979). Power analysis can be used to evaluate the robustness of the test data to detect effects at one point in time or to distinguish trends and directions in a time series of data.

3.3.5. *Methods for Evaluating Large-Scale and Long-Term (Regional) Effects*

Because of the incomplete understanding of how to aggregate from species and local-scale effects to long-term regional effects, information and guidance is still developing and, for the time being, must be used provisionally through case studies. Probably the best documented case, reflecting progress in methods but also problems in the assessment of causal mechanisms, is the work done on understanding changes in Great Lakes fisheries (Eshenroder et al., 1991). A key issue is whether the decline in traditional, self-reproducing fisheries is attributable to overexploitation of these top predators, to enrichment (and related food chain effects), or to the regional toxic burdens acting on sensitive stages in reproduction. All of these alternatives are explored in the 1985 review of the Great Lakes Water Quality Agreement (NRC/NAS/RSC, 1985), and all are judged to be contributors in different degrees in each of the major lakes. Determining the stressor magnitude was difficult in each case, as was documenting either a theoretical or empirical causal mechanism for the effects: impairment of stock replacement by overexploitation; impairment of reproduction through toxic effects; or change in aeration of the cold bottom-water habitat attributable to increased decomposition demand following enrichment (NRC/NAS/RSC, 1985). Risk assessment is obviously hampered when even detailed retrospective studies have difficulty attributing outcomes to one or another causal mechanism operating on a large scale.

Another example that is still being debated is the regionwide effect of acid rain and oxidants on U.S. forests, especially in the Northeast, the Southeast, and in southern California (Shriner et al., 1990). The problem is affected by the multiplicity of pollutants involved, differences in dose, great differences in species sensitivities, the evidence of previous declines (which may or may not be due to a quite different "cause"), and the great difficulty in extrapolating from seedling tests to long-term whole-tree effects. Regionwide effects characterization also is limited by the absence of a consensus on methods for landscape-scale effects (Sheehan, 1994).

One final example of an evaluative approach at a large scale is that developed through the EPA/Science Advisory Board (SAB) study of priority problems, the most serious apparently being large-scale, large-impact, high-uncertainty problems that are not readily quantified (Harwell et al., 1992).

3.3.6. *Balancing Local-Scale/Small-Risk Problems With Good Data in the Same Assessment With Large-Scale, Potentially Large-Risk Problems Having Poor and Insufficient Data*

The approaches described in previous sections indicate a methodology in which we have some confidence when the stressor is a single, known chemical acting on defined life stages of a known species or population. The tributyltin case study is a useful example (NRC/NAS, 1993). This and five other case studies at larger scales are illustrated in table 5-4. The challenge in large-system risk assessment, however, frequently requires that we consider multiple chemicals (as in the Great Lakes) circulating over a large area and potentially affecting many species in a wide array of poorly known ecosystem functions. Changes in systems at these scales obviously affect many more people and their commerce than most single-species effects, although large-system effects mostly begin with single-species effects.

When the multispecies system is poorly understood, we do not know whether single-species information can be combined satisfactorily through large-area or long-term averaging. One obvious option is to consider refraining from performing a risk assessment when the uncertainties inherent in large systems are apparent. Does one then treat that part of the system for which data are available as the major component of the effects characterization problem? Probably not if the risk assessment is to have credibility. It is possible that the mechanistic approach is inappropriate, for reasons discussed by Dowlatabadi and Morgan (1993) in relation to assessments of the risks from climate change.

The principal institutional response for situations where long-system assessments may not be possible should be to seek new, appropriate data, often for large areas and/or long periods. Some assessments may have to be postponed or methods may have to be developed using broad judgments of potential risks, at least until better information and better tools become available.

Table 5-4. Scales of Observation and Management in Case Studies Evaluated by the NRC/NAS Committee (NRC/NAS, 1993)

	Observational Scale		Management Scale	
Case Study	**Spatial**	**Temporal**	**Spatial**	**Temporal**
Tributyltin	‹ 1 m^3 ~ 1 ha	‹ 1 yr (laboratory) ‹ 5 yr (field)	Chesapeake Bay	› 5 yr
Agricultural chemicals	~ 1 ha	‹ 1 yr (field)	Agricultural region	› 5 yr
PCB and TCDD	‹ 1 L	~ 1 mo (laboratory)	Lakes or rivers	› 10 yr
Spotted owl	~ 300 km^2	‹ 6 yr	Pacific Northwest	› 100 yr
Species introduction	‹ 100 m^2	~ 1 yr (greenhouse)	Agricultural region	› › 1 yr
Georges Bank	~ 10^4 km^2	Last 30 yr	~ 10^4 km^2	Next 5 yr

4. ECOLOGICAL RESPONSE ANALYSIS: ISSUES OF MEASUREMENT

As outlined in the ecological risk assessment framework (chapter 1), "ecological response analysis" is the most central concept in effects characterization. In this section, we consider issues of measurement in response to chemical and physical stressors. Central to this discussion is the problem of understanding and measuring direct versus indirect effects of chemical and physical stressors on individuals, populations, and ecosystems. In the next two sections, we consider the role of modeling and of different types of endpoints.

4.1. Chemical Stressors

The effects of chemicals on individual organisms, and to a lesser extent on populations, have been widely studied. As a result, we are able to be precise as to what constitutes measurement of a known direct effect and what represents measurement of an indirect effect.

4.1.1. *Measuring Direct Effects of Chemical Stressors*

The effects of substantial exposures of a variety of chemicals on sexual maturation and gamete development have been reported for fish (Donaldson and Scherer, 1983), amphibians (Martin, 1983), and invertebrate taxa (Dixon, 1983; Davey et al., 1983). Reduced egg production, reduced hatch success, and the reduced size of brood per female and the viability of hatchlings have been documented for substantial chemical exposures to birds (e.g., Peakall, 1983), fish (e.g., Birge et al., 1979), and aquatic invertebrates (e.g., Hatakeyama and Yasuno, 1981). In field studies, Schofield (1976) observed a reduction in the reproductive success of freshwater fish in low pH lakes.

A variety of laboratory toxicity tests and field studies in highly contaminated environments have documented increased (above normal) mortality rates for a variety of vertebrate and invertebrate species at specific chemical exposures. These data have been reviewed on a chemical-specific basis by the EPA for ambient water quality and health assessment documents, and by the U.S. Fish and Wildlife Service (U.S. FWS) for contaminant hazard reviews.

A variety of field studies have shown reductions in the abundance of exposed populations to be inversely correlated with increasing chemical contaminant concentrations; selected studies were reviewed by Sheehan (1984a, b, 1991). Reductions in population abundance and the complete extinction of sensitive populations resulting from the stress of chemical exposures are the primary factors altering population structure. Such effects may not be entirely attributable to toxicity effects, but may be due also to a direct reduction in reproductive success (rather than death of adults) or the

inability of the organisms to function successfully in competitive interactions. One of the better documented trends in population extinction was recorded for fish during the period of industrially stimulated acidification of poorly buffered lakes (e.g., Muniz and Leivestad, 1980).

Within exposed fish and wildlife populations, the measured incidence of various nonspecific disease symptoms appears to be associated with chemical exposures in degraded aquatic and terrestrial environments. Sindermann et al. (1980) reported an increased incidence in ulcers and fin rot in fish from highly contaminated estuarine waters. Hetrick et al. (1979) measured susceptibility of rainbow trout to an infectious virus after exposure to sublethal levels of copper. A similar reduction in tolerance to disease and insect infestations has been reported for chemically exposed plants (e.g., Treshow, 1978).

4.1.2. *Measuring Indirect Effects of Chemical Stressors*

Several major studies, but especially the acid-deposition research sponsored by the National Acid Precipitation Assessment Program (NAPAP), have begun to document the importance of indirect effects on organism metabolism largely mediated through changes in water or soil chemistry. An interesting example is induction of the toxic responses of fish to monomeric aluminum in lakes receiving acidity from rainfall (NAPAP, 1989a). The higher hydrogen ion (H^+) concentration can induce ionic effects on organisms, but the more profound effect on fish is an indirect one, a result of inducing a new stressor in the form of solubilized aluminum, an infrequent, often storm-related phenomenon.

Direct and indirect effects of air pollutants on forests are illustrated in table 5-5. Most of the direct effects (e.g., altered seed production, reproduction, nutrient cycling) also can be seen as a prospective indirect effect from the pollutant's direct effect on foliage. A secondary indirect effect at the ecosystem level is possible through the change in seed production, which leads to changes in species composition or changes in photosynthesis, which leads to reduced standing biomass and increased or decreased insect and disease epidemics (depending on the species). These results illustrate that the very concept of what is direct and what is indirect needs to be treated carefully.

Another example of a direct effect leading to an indirect effect is evident when the acidification of soil horizons (through H^+ addition) leads to losses of Ca and Mg from the soil nutrient pools and the immobilization of P and solubilization of aluminum (Al) at low pH levels. In this example, the hydrogen ion concentration, or change in H^+, is not itself the proximate cause of P deficiency or Al toxicity in plants, but the means by which these new stressors and a different exposure regime is

Table 5-5. Interaction of Air Pollution and Temperate Forest Ecosystems Under Conditions of Intermediate Air Contaminant Load—Designated Class II Interactions (Smith, 1981)

Forest Soil and Vegetation: Activity and Response	Ecosystem Consequence and Impact
1. Forest tree reproduction, alteration, or inhibition	1. Altered species composition
2. Forest nutrient cycling, alteration a. Reduced litter decomposition b. Increased plant leaching, soil leaching, and soil weathering c. Disturbance of microbial symbioses	2. Reduced growth, less biomass
3. Forest metabolism, alteration a. Decreased photosynthesis b. Increased respiration	3. Reduced growth, less biomass
4. Forest stress, alteration a. Phytophagous insects, increased or decreased activity b. Microbial pathogens, increased or decreased activity c. Foliar damage increased by direct air pollution influence	4. Altered ecosystem stress a. Increased or decreased insect infestations b. Increased or decreased disease epidemics c. Reduced growth, less biomass, altered species composition

induced (Shriner et al., 1990; Loucks et al., 1993). Obviously, the mechanisms by which indirect effects are expressed are difficult to discern, and they are quantifiable only through intensive studies of the mechanisms involved.

Some alteration of ecosystem properties also can be induced indirectly through changes in the chemical environment of organisms functioning in decomposition processes. Food quality for these organisms can be changed, inducing changes in the mix of species (populations) making up the decomposition community, in both aquatic and terrestrial environments.

Two recent studies indicate that insecticide-induced invertebrate mortality in sprayed areas may indirectly affect birds by reducing the abundance of available prey organisms and destablizing their food web (Sheehan et al., 1987, 1994a; Pascual and Peris, 1992). In both cases, synthetic pyrethroid insecticides that are highly toxic to aquatic and terrestrial macroinvertebrates, but pose no toxic hazard to birds, were used in insect pest control, resulting in substantial nontarget invertebrate mortality. The insecticide-induced shortage in prey availability was shown to be particularly critical when the birds are obligate feeders on macroinvertebrate prey.

Herbicides also have been shown to affect bird populations indirectly by reducing available nesting habitat and protective cover. Dwernychuk and Boag (1973) reported a 74 percent reduction in the nest density of ducks in an area sprayed with 2,4-dimethylphenol. In contrast, Rands (1985) demonstrated how cereal grain-field headlands left unsprayed with herbicide supported larger grey partridge broods than sprayed headlands. The success in unsprayed areas may be an indirect outcome of the improved cover afforded by the increased vegetation, which protects the partridge broods from discovery by predators.

4.2. Physical Stressors

Experience in measuring physical stressors, either direct or indirect, as agents in risk assessment is more limited than with chemical stressors. The important factors for which data exist include changes in the hydrology of a site, the introduction (or absence) of fire, and the physical manipulation of forests, fisheries, or wildlife populations through harvesting. All of these treatments can induce a wide range of effects, both direct and indirect.

Probably the most widespread are the direct effects on organism metabolism, populations, and communities induced by changes in soil-water availability or flows in streams and wetlands. The facilitation of runoff following land conversions, induction of water-level change in wetlands, and associated erosion and sediment transport are among the most visible direct effects acting themselves

as stressors (Loucks, 1990a). A special case of hydrologic effect is evident in the changes induced by altering the normal amplitude and seasonality of peak water level and drawdown in swamps and marshes (Bedford and Loucks, 1984; Loucks, 1990b). Although in these examples, as with other physical stressors, a dose-response relationship is implied, more experience is required in adapting physical system measurements into a form equivalent to dose for risk assessment purposes.

The periodic combustion of accumulated growth in terrestrial ecosystems also must be considered a physical stressor, just as flood intensity is a stressor in aquatic ecosystems. Human activities change the frequency of both events, thus altering characteristics of the ensuing recovery transient and the status of areawide trajectories or destabilizations. The direct effects that ensue from changes toward increased or decreased frequency of fire (e.g., in Yellowstone National Park or in the West Coast chaparral ecosystems) produce a wide range of indirect effects on species through change in the severity of the fires, change in the potential to maintain diverse habitats, and alterations of other processes.

Like fire, the physical removal of forest mass (e.g., clear-cutting, chipping) or the physical altering of habitat or the induced dominance of selected species by chemical treatments must be viewed as a physical stressor leading to observable ecological effects, both short term and long term. These direct effects may be more important for the indirect effects they facilitate (e.g., erosion and changes in nutrient cycling) than the physical exposure itself. Many of the resulting indirect effects are well documented for common forest ecosystems (Bormann and Likens, 1979), but poorly documented for rare ecosystem types containing unusual biological diversity. The long-term effects of harvesting on the composition of the herbaceous understory (e.g., Duffy and Meier, 1992) is not well known for any forest ecosystem.

In aquatic ecosystems, physical manipulations include the altering of flow regimes in rivers, lakes, and reservoirs. A good example of effects that should have received a risk assessment are those resulting from reservoir management, as documented in the Glen Canyon Study (NRC, 1987). Here the type of outflow (clear and cold bottom water) induces major changes in the Colorado River below the dam until sediment resuspension and temperature return to normal, a process that itself changes long stretches of the river. Other indirect effects from these physical stressors also are interesting: the characteristic flushing times of lentic and coastal environments are changed, often affecting the seasonal reproductive pattern of aquatic species. Hypolimnion aeration, key components of decomposition, and nutrient cycling all can be affected, leading to new system dynamic responses

or destabilization of periodic specialized habitats within otherwise lightly impacted systems (NRC/NAS/RSC, 1985).

4.3. Multiple Stressors: Aquatic Ecosystem Examples

In aquatic ecosystems, populations are often exposed to a mixture of chemicals in water and sediments, to combinations of physical and chemical stressors, and to chemical stressors and human exploitation pressures. Approaches to characterizing the effects of multiple stressors and to quantifying the relative contribution of individual stressors to the aggregate population response are being developed.

4.3.1. *Multiple Chemicals in Aquatic Systems*

Most effluent released to aquatic systems contains a variety of chemicals and the sediments in the waterways in industrial areas contain a variety of metals and persistent organic substances. The occurrence of chemicals in mixtures influences toxicity in two ways: (1) chemical mixtures can cause a toxic effect that is qualitatively or quantitatively different from any component acting alone; and (2) the effects of one chemical may influence the kinetics of uptake, metabolism, or excretion of other chemicals. An effect measured for organisms exposed to multiple chemicals is an index of the response of the organisms to the aggregate mixture. The most direct way to determine toxicity is to test the mixture directly. Based on whole-effluent, receiving-water, or sediment toxicity tests alone, however, no conclusions can be drawn as to the contributions of specific chemicals to the measured toxicity; similarly, single-chemical toxicity test data alone are insufficient to predict accurately the toxicity of a chemical mixture.

The most commonly used model of joint toxic action is the concentration addition model (Finney, 1971; Marking and Dawson, 1975). This model is based on isobole theory and involves the use of the toxic unit concept to sum the action of various components of the mixture according to the formula:

$$\sum C_i / LC_{50_i} = 1$$

where:

C_i = the concentration of chemical i
LC_{50_i} = the LC_{50} value for chemical i

The ratio C_i / LC_{50_i} is termed the toxic unit. This model is applicable if chemicals act on similar sites with similar modes of action. A second model, the response additive model, assumes that the site and modes of action of the chemical are different and the chemicals do not interact. Thus, the organism responds to the chemical to which it is most sensitive, relative to the chemical's concentration (Kodell and Pounds, 1985). This relationship is expressed as:

$$P(C_1 + C_2) = P(C_1) + [1 - P(C_1)]P(C_2)$$

where:

$P(C_1)$ = the probability of a response, given concentration C_1

$P(C_2)$ = the probability of a response, given concentration C_2

In retrospective studies of effluent and sediments, the toxicity identification evaluation (TIE) approach described by EPA is useful in quantifying the contribution of constituents of the mixture to toxicity (U.S. EPA, 1988c, d, e). The TIE approach (1) either separates components from the mixture or masks their activity (e.g., addition of chelators to bind metals), (2) retests the toxicity of the amended samples, and (3) compares the results with those for the original mixture.

In recent years, scientists have used TIE methods to identify compounds responsible for toxicity and to determine the relative contribution of specific chemicals to observed toxicity in ambient water and aqueous fractions of sediments (e.g., Ankley et al., 1990; Schubauer-Berigan and Ankley, 1991; Ho et al., 1992; Balch et al., 1993; Kline et al., 1993; Ogle et al., 1993). Methods for conducting bulk sediment TIE evaluations are being developed (Ho et al., 1993).

4.3.2. *Chemicals and Human Exploitation of Aquatic Resources*

One topic that has received little attention is the effect of chemical (or for that matter, physical) stressors on aquatic populations that are heavily exploited for human use. The issues are twofold. First, does chemical accumulation limit the fisheries resource for human use? Second, does the effect of chemical exposure in conjunction with the stress of harvesting (and other stressors) pose a threat to the populations?

The EPA national study of chemical residues in fish for 106 sites showed that polychlorinated biphenyls (PCBs) at 42 of these sites posed an estimated lifetime cancer risk of 1 in 10,000 to human consumers of fish (U.S. EPA, 1992b). Of equal concern is the effect of such chemical exposures on the sustainability of a harvested fishery. In theory, because high existing levels of stress reduce the

ability of populations to resist new stress regimes, exposing a heavily harvested fish population to a chemical at a toxic concentration would further reduce the population's abundance and/or recruitment and lower its probability of survival. A corollary to this theory is that increasing the rate of harvesting should increase the sensitivity of a population to a given chemical exposure. Although this theory is widely held, few quantitative studies have estimated the relative magnitude of effects of chemical and harvesting regimes on fisheries. The Great Lakes studies described earlier in this section provide one example. Another is provided by Barnthouse et al. (1990), who evaluated the interaction between fishing mortality and chemical toxicity in striped bass and menhaden populations using population models. The study confirmed the increased sensitivity of heavily exploited populations over less-exploited populations to chemical exposures. The relative effects of chemicals and harvesting also has been examined for duck populations (Sheehan et al., 1987).

4.4. Multiple Stressors: Terrestrial Ecosystem Case Study

The characterization of effects resulting from combinations of natural and anthropogenic stressors is a common but difficult problem. The direct and indirect effects often operate on multiple species and multiple ecosystem processes. The problem may be illustrated best by current research on the effects of acid deposition and O_3 on poorly buffered oak forests in the Ohio Valley (Loucks et al., 1993). Earlier studies, with experimental support, had proceeded on the expectation that a base-titration reaction in the soil would take place over some very extended period of time, up to a century (Shriner et al., 1990). The result would be a loss of cations and nutrient depletion, possibly affecting tree growth. This example concerns one stressor, acidity, acting by itself and inducing other stressors (including the possibility of periodic aluminum toxicity). This multiple-stressor situation can then also combine with a natural stressor such as drought.

Testing hypotheses as to the patterns of effects on forests from acidic inputs must start with documentation of the dose, both current and cumulative historical ion-deposition amounts, and the concentration of pollutants (e.g., O_3) that may act over similar time spans. The data for this study were obtained along a gradient in sulfate (SO_4^{2-}) deposition from Illinois to Ohio (the Ohio Corridor Study), with funding from the U.S. Forest Service from late 1986 to 1990 (Loucks et al., 1993). The results show even steps along the gradient between each of the three states for cumulative deposition of SO_4^{2-} and NO_3^{1-} during the past 8 decades, ranging from 6.9 eq/m^2 in Illinois to 9.6 eq/m^2 in Ohio for only the period between 1900 and 1949. Other results show that soils derived from similar sandstone substrates differ along the gradient in several ways that are indicative of acidic deposition

effects: The eastern site (Ohio), with the longest period of high deposition and the greatest cumulative dose, is characterized by significantly lower pH (3.95) and a higher percentage of carbon in the surface mineral (Al) horizon than the lower-dose western sites (pH 4.57). The lower-mineral B horizon of the high-dose site in Ohio shows significantly lower pH, lower total bases, and a lower percent base saturation than the reference site in Illinois (Loucks et al., 1991). The lower level of bases on the high-dose sites is proportional to estimates of 50- and 86-year cumulative wet-plus-dry deposition of SO_4^{2-} and NO_3^{1-}, and to the higher level of exchangeable Al^{3+} at the eastern site. The Ca:Al ratio also is significantly lower at the eastern high-dose site. Thus, in addition to the O_3 and strong ion inputs, other stressors have been induced.

The soil fauna should be one endpoint complex to be evaluated for effects, given that they are short-lived species closely linked to soil chemistry. These show greatly reduced populations in the higher-dose Indiana and Ohio sites compared with southern Illinois. Earthworm populations are $1/m^2$ compared with $30/m^2$ at the reference site. An exacerbation of these differences is evident in the year following a severe drought (i.e., 1989) compared with 1990. Tree growth, a different endpoint potentially responding to either or both Al^{3+} and O_3 dose, was evaluated through the individual-tree basal area increment (BAI) from 1934 to 1960 and 1960 to 1987 (before and after the most serious elevation of pollutants). The BAI change shows a decline at sites with low soil Ca:Al ratios, while little or no decline is evident at sites with moderate Ca:Al ratios. Tree mortality rates, a third measurement endpoint complex, is significantly higher during the decade 1978 to 1987 in comparison with 1968 to 1977, despite little difference in weather. This response could be due primarily to the O_3 stressor, which is uniformly serious across all sites.

This example illustrates that with sufficient data on a variety of stressors and with a reasonable knowledge of probable mechanisms, it may be possible to attribute different kinds of effects to different stressor regimes. A similar approach should be possible in streams where both siltation from nonpoint source pollutants and periodic inputs from agrichemicals stress the same system but probably produce somewhat different specific as well as joint effects.

5. ECOLOGICAL RESPONSE ANALYSIS: THE ROLE OF VARIOUS TYPES OF MODELS

Models are an important part of the effects characterization process, although many potential uses of models have not been fully realized. In effects characterization, models may be used for prediction, explanation, or extrapolation. The principal use of models in effects characterization, however, is to describe the relationships between the duration and intensity of the stress regime and the response of the exposed biological system. Although numerical models are important tools in effects characterizations, the validation of numerical models for natural systems is impossible (Oreskes et al., 1994). This is because natural systems are not closed and model results are already non-unique. Although these models cannot be radiated, they can be confined by the demonstration of agreement between observation and prediction.

5.1. Types of Models

Three types of models are used in effects characterization: physical, statistical, and mechanistic. Physical models are material representations of the system that can be manipulated or tested. Single- or multispecies toxicity tests are commonly used physical models in effects characterization. Physical-toxicity test models and field-study models of biological effects provide much of the current exposure-response data for effects characterization.

Statistical models attempt to derive generalizations by using regression and other statistical techniques to summarize experimental stress regime-response data. Concentration dose-response models are obtained by statistically fitting a continuous function such as a probit or logit to results of toxicity tests with discrete concentrations or doses. The model assumes that the response of exposed organisms to the chemical exposure can be characterized by a statistical distribution independent of the mechanism of toxic action. Statistical regression models can be used to quantify the relative sensitivity of different taxa or life stages to stressors (e.g., Mayer and Ellersieck, 1986). Multidimensional response surface models have been used to quantify interactive effects of various stressors such as chemicals and pH on biota (e.g., Mount et al., 1988). All statistical models are strictly empirical approaches for fitting curves and surfaces to test data.

Mechanistic models describe the relationship between some phenomenon and its underlying cause in quantitative terms. Types include pharmacodynamic or toxicodynamic models that incorporate mechanisms of uptake, distribution, and toxic action. In principle, mechanistic models can account for characteristics of organisms (e.g., age, nutritional status, reproductive status) that are known to influence sensitivity to stressor exposure. The proper accounting for individual or population

characteristics would provide assistance in extrapolating test data from species to species and from laboratory to field conditions. The greatest benefit to improving stress regime-response characterization is expected in the development and application of mechanistic models.

Four reviews of the models available for assessing the ecological risks of toxic chemical exposures have been published in recent years (Barnthouse et al., 1986; Barnthouse, 1992; Emlen, 1989; Suter, 1993). Organism-level effects models include toxicodynamic models (Mancini, 1983; Kooijman and Metz, 1984; Lassiter, 1985; Lassiter and Hallam, 1990) that relate the risk of mortality to the uptake and internal concentration of chemicals. Models of the effects of stressors on energetics and growth (Sugden and Harris, 1972; Kitchell et al., 1977; Rice et al., 1983; Bartell et al., 1986) also are relevant to effects characterization but have not been extensively applied to date.

The population-level models of current or potential use in risk assessment include the many models developed for management of fish and wildlife populations. These are primarily demographic models that predict the effects of physical, chemical, or biological stressors on population parameters such as abundance or recruitment. Emlen (1989) suggests that the most relevant endpoints for population modeling are "pseudo extinction"—an abstraction that measures the probability of a population falling below some predetermined fraction of its undisturbed value—and alteration of the temporal mean population abundance. Logan (1986) and Barnthouse et al. (1987, 1989, 1990) provide examples of the use of fisheries-derived models in chemical risk assessment. Barnthouse et al. (1987) developed a method for extrapolating logistic functions and confidence limits for untested fish species. The study used mortality data from three life stages (egg larvae and juveniles) to encompass the fish life cycle from egg to first reproduction. The reproductive potential for a 1-year-old female recruit then was modeled using a form of demographic population modeling that accounted for the female's annual probability of survival at different ages, her expected fecundity at different ages (provided that she survives), the probability that a spawned egg will hatch, and the probability that a newly hatched juvenile will survive to age one. Also, a few examples are available of the application of population models to characterize the effects of chemicals on wildlife. Grant et al. (1983) evaluated the effects of vertebrate pesticides on great horned owls using a Leslie Matrix model approach. A similar approach was used to predict the recovery of seabird populations from the stress of an oil spill (Samuals and Ladino, 1983). Tipton et al. (1980) used a different approach, classifying bobwhite quail into various categories, with respect to their physiological response, to characterize the effects of contamination by methylparathion.

Community and ecosystem models are the most complex and diverse among ecological effects models. They may be site specific (Kremer and Nixon, 1978; Andersen and Ursin, 1977) or generic (O'Neill et al., 1982); generally they are designed to mimic portions of ecosystems. The driving variables are physical and chemical factors (e.g., temperature, pH, nutrient concentration, chemical concentrations), and population interactions (e.g., grazing or predation) are described with mathematical relationships. The effects characterization component of these models expresses the effect of the stress regime on the parameters of interest in the exposed ecological system. Large-scale models for regions or landscapes have been proposed to characterize ecological effects (Dale and Gardner, 1987; Graham et al., 1991), but such models have not yet been widely used in risk assessment.

5.2. Chemical-Concentration/Dose-Response Models

The importance of toxicity tests for characterizing effects for individuals and subsets of populations is noted in section 3. Test results are interpreted with chemical-concentration/dose-response models. For chemicals, the concentration/dose-response relationship is a graded relationship between the concentration or dose of the chemical to which the organism is exposed and the severity of the response elicited. Generally, within certain limits, the greater the concentration or dose of the chemical, the more severe the response. The curve fitted to represent this relationship generally will be asymptotic because at all concentrations below some minimum threshold value no measurable adverse response will be elicited, while at all concentrations above some maximum value, most or all of the organisms will be adversely affected. The steeper the slope in the central portion of the curve, the more intense the response over a narrow range of concentration or dose. For the concentration/dose-response relationship to have precise meaning, the duration of exposure must be specified.

5.2.1. *Concentration/Dose-Response Function*

The most common two-dimensional model in ecotoxicology is the concentration- or dose-response function (figure 5-4). In this model, although time is not included, response data are for a specified duration of exposure, and either severity or proportion responding is eliminated in order to have a single response variable. Normally the percent response is plotted against concentration or dose. The probit, logit, Weibull, or other function is fitted to the discrete concentration/dose-response data from the toxicity test to provide a continuous response model for the range of exposure. This function is

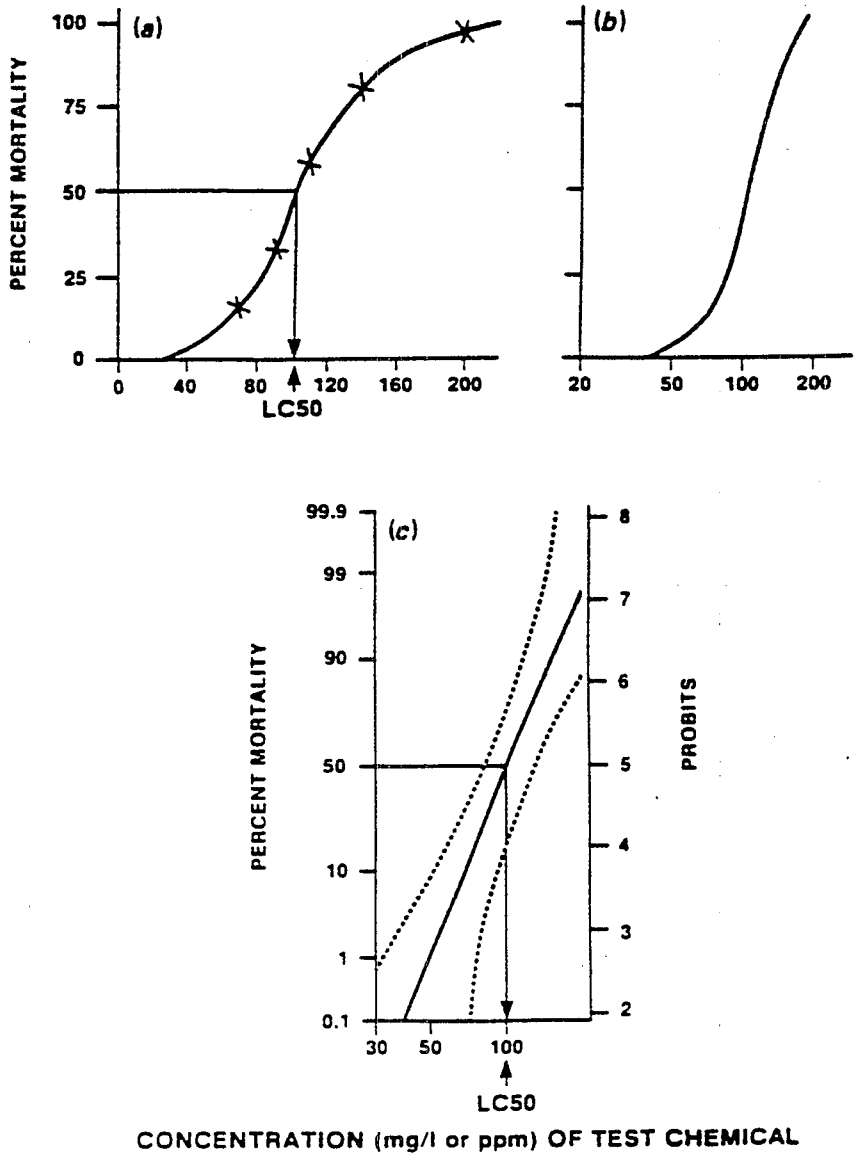

Figure 5-4. **Mortality in a fish population exposed to a range of concentrations of a chemical in water (from Rand and Petrocelli, 1985):**

- **(a) percent mortality versus concentration plotted on an arithmetic scale;**
- **(b) the same data as in (a) but with mortality on an arithmetic scale and concentration on a logarithmic scale;**
- **(c) the same data as in (a) but with mortality expressed as probits versus concentration on a logarithmic scale (the dotted lines on each side of the curve represent the 95 percent confidence limits).**

used to generate effect concentrations (e.g., EC_{10}, LC_{50}, LD_{50}). It is important that the concentration/dose-response curve be presented, rather than merely an LC_{50} or LD_{50} estimate, to provide a complete interpretation of the test results.

Time-response functions are like concentration-response functions but are used to generate an LT_{50}. Time-response functions are useful when the concentration is relatively constant but the duration of exposure is variable.

A two-dimensional model that includes time is the time-concentration function. This model is created by collecting data at multiple times during the toxicity test and calculating an EC_{50} or LC_{50} for each time. These concentrations are then plotted against time and a function is fit to the data. The resulting curve may indicate when acute lethality has ceased—in places where the curve is asymptotic to the time axis. The LC_{50} for the time that is in the asymptotic part of the curve is termed the threshold or incipient LC_{50}.

Although three-dimensional representations of toxic effects are rare, they provide an effective way to present toxicity test data as a response surface (figure 5-5). Such models have the obvious advantage of allowing the risk assessor to estimate the level of effects from combinations of intensity and duration of exposure.

5.3. Probabilistic Models to Handle Uncertainty

Human health risk assessment models have used Monte Carlo simulations to evaluate uncertainties in both exposure and dose-response assessments (e.g., McKone and Bogan, 1991). Similar models are now being used in ecological risk assessments (Bartell et al., 1992; Sheehan et al., 1994b). Probabilistic models incorporate distributions of concentration/dose and effect parameters from laboratory and field tests and use Monte Carlo simulation of the curve-fit model to generate a probability distribution of the concentration/dose-response relationship. This method describes the range of possibilities and the frequency with which any one possible form of the response might occur.

The value from increased use of probabilistic techniques in describing concentration/dose-response relationships is in the incorporation of test uncertainties into the description of the relationship.

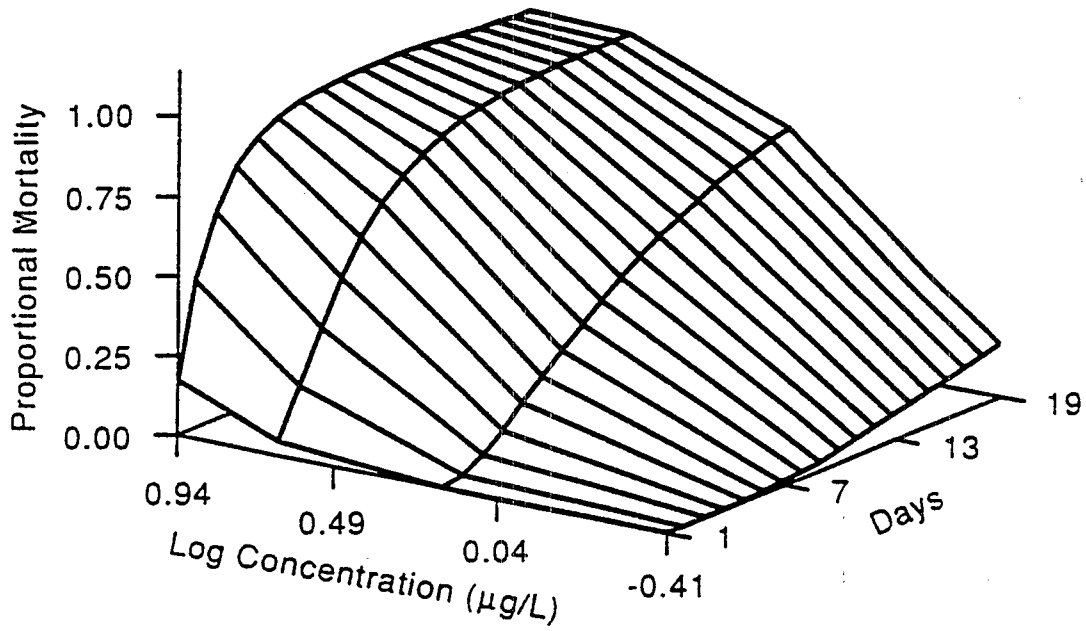

Figure 5-5. Toxic effects as a function of concentration, duration, and proportion responding (from Nimmo et al., 1977).

5.4. Approaches to Stressor-Response Measurement and Modeling for Ecosystems

Stressor-response modeling for the ecosystem area of ecological risk assessment will require greater attention if our aspirations to predict effects at higher levels in the biological organization are to be realized.

5.4.1. *Concentration-Response Models for Ecosystems*

Few attempts have been made to model chemical effects on population and ecosystem endpoints. The Standard Water Column Model (SWACOM) provides an example of such a modeling approach (O'Neill et al., 1982). SWACOM presents a Monte Carlo simulation approach for extrapolating laboratory toxicity data to predict effects such as decreased productivity and reduction in fish biomass. The translation for toxicity data incorporates the information on the mechanism of toxic action, with population interactions taken into account and the uncertainty in laboratory measurements retained. Risk estimates are generated in the form of probabilities that an effect could occur. A detailed description of ecosystem simulation modeling is provided in Bartell et al. (1992).

The strengths of this methodology are that a variety of endpoints can be assessed for each source-exposure regime and the risk algorithm can be applied to a variety of chemicals. Such simulations provide information on the range of possibilities considered by the predictiveness of the model for a specific ecosystem. Their usefulness can only be judged by comparing field observations with the predictions. Further application and refinement of such models will depend on refinements in our understanding of the mode of toxic action of the chemicals of interest and of the ecological interactions within the system to be simulated.

5.4.2. *Models of Cascading Effects in Aquatic Ecosystems*

Some of the most complex and counterintuitive responses of ecosystems to external manipulation are being modeled successfully in clear, low-nutrient northern lakes (Vanni et al., 1992). Freshwater plankton communities in these lakes are regulated by a variety of factors, among which nutrient enrichment and predators are two of the most important. Both are often manipulated by humans, and can, therefore, be thought of as anthropogenic stressors. An increase in limiting nutrients can stimulate production and biomass of phytoplankton, which in turn can stimulate production and biomass of herbivorous zooplankton. Predators such as fish can influence plankton communities through selective predation on large versus small zooplankton species. Because large zooplankton have relatively high grazing rates (per individual) and graze on a wider range of food particles, size-

selective predation by fish on large zooplankton also can have a substantial influence on phytoplankton and nutrient cycling. The series of opposing interactions illustrated in figure 5-6 have come to be known as "cascading effects" because of the way control from above is expressed stepwise down through the system to primary producers.

The influences and interaction among these factors have been investigated quantitatively through manipulation experiments and simulation modeling (Vanni et al., 1992). The modeling has been used to forecast the responses to planned manipulations and to elucidate the mechanisms responsible for changes induced in response to altered predator and/or nutrient conditions. Scavia et al. (1988) had used simulation modeling earlier to investigate the influence of food web interactions on the magnitude and scale of variability in primary production in Lake Michigan, with good descriptive results.

The work by Vanni et al. (1992) used simulation modeling to investigate various scenarios of planktivory and phosphorus-loading rate and used observed responses in Lake Mendota phytoplankton to test the effects of those scenarios. The model, which has obvious hierarchical control features and feedbacks, simulates the Lake Mendota plankton food web of the late 1980s and accurately predicts many consequences of observed food web changes. The results suggest that food web manipulation represents a viable strategy for reducing the severity of summer algal blooms. Vanni concludes that simulation models such as the one he tested can be useful tools in forecasting the consequences of food web manipulations and thus are illustrative of tools available for characterizing ecological effects in risk assessment.

In terrestrial systems, the competition models (for nutrients), and their outcomes for species as developed by Tilman (1989), provide a foundation for exploring both direct and indirect effects. The models are potentially applicable at population, community, and ecosystem levels. In addition, these models have been adapted by Landis (1986) to evaluate the impact of xenobiotics on biological communities.

5.5. Lake and Forest Ecosystem Component Models Developed in NAPAP

One of the largest investments in large-system stressor-response modeling was that undertaken as part of NAPAP between 1982 and 1990. Two reports summarize those results in one place (NAPAP, 1989b, 1990), but some of the results are published in numerous other papers as well. Watershed and aquatic ecosystem models likely to be useful in risk assessment include the following: Integrated Lake-Watershed Acidification Study (ILWAS), for northeastern mountain lake ecosystems; Model of

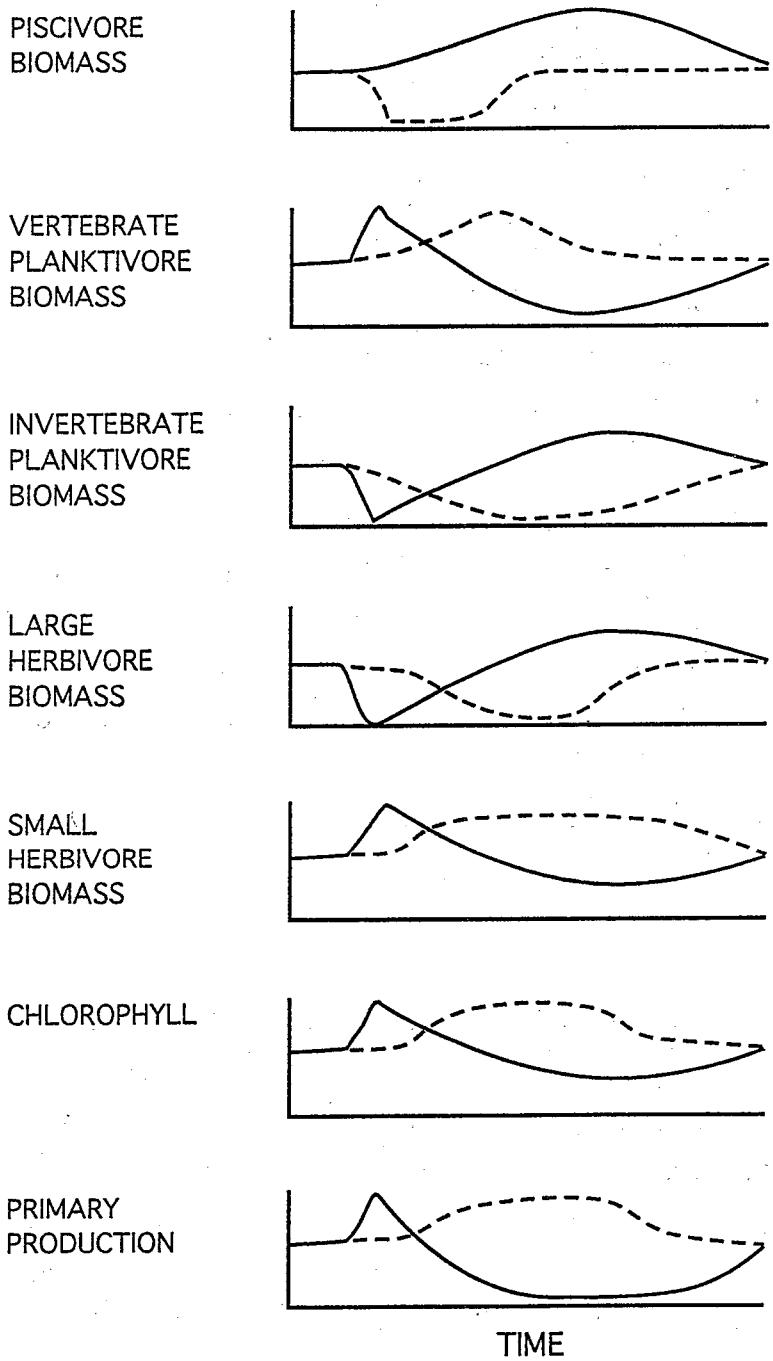

Figure 5-6. Time-course of ecosystem response to a strong piscivore year class (solid line) and a partial winter kill of piscivores (dashed line) (Carpenter et al., 1985).

Acidification of Groundwater in Catchments (MAGIC), for soil chemical, ground-water, and stream chemistry changes; and the Regional MAGIC Model for analysis of areawide changes in stream chemistry. Questions of error propagation, validation, and appropriate applications for these models have all been discussed at length in the NAPAP reports.

An even larger number of models were developed and tested for understanding forest responses, but most only concerned individual components of ecosystems. Important among these are the following: Nutrient Cycling Model (NuCM), useful in completing mass balances of nutrients at specific sites and in simulating long-term trends from anthropogenic deposition on forest vegetation and soils; and Response of Plants to Interacting Stresses (ROPIS), a useful model developed at Cornell University for evaluating the interactions between canopy air pollution effects and soil-water and nutrient uptake and growth. Both oxidant and acidity effects can be evaluated.

These examples of relatively well-documented, large-scale stressor-response models are cited to illustrate that although substantial computational capability exists for assessment of risks from a range of external stressors, the local, species-specific, and site-specific data requirements are substantial—something that should be recognized. A handbook of species and ecosystem model parameters drawn from published papers would probably be very helpful in developing a generalized, but quantitatively precise, "expert-system" approach to ecosystem risk assessment.

5.6. Hierarchical Approaches Integrating Individuals, Populations, Subsystems, and Ecosystems

In situations where the known stressor is operating at the level of individual organisms (as opposed to the mostly mass balance or physical system-level effects discussed above), but the endpoint of interest is at the ecosystem level, a somewhat different approach to system integration across scales may be required. For example, work by Cai et al. (1993) breaks up the processes of forest litter decomposition into a hierarchy of soil faunal feedbacks and species demographics. An array of linked models is being developed—from individual-based to ecosystem level—to incorporate the effects of rain chemistry inputs and soil pH changes (described earlier) on soil fauna. The individual-based model is used first to incorporate known effects of high H^+ concentrations on the sensitive life stages of earthworms and other macroinvertebrate groups. Dose-response relationships for individuals can then be used to compute the changes induced in populations of some of these species, which as noted in section 4.4 have experienced up to a 90 percent decline in numbers at high-dose sites in the Ohio Valley (Loucks et al., 1991). The consequences of these changes for the

mass action of decomposition (a rate change) can then be computed for the forest ecosystem as a whole. The model is being validated against the soil nutrient pool data obtained in the field.

5.7. Evaluation of Causality

The difficulty in appropriate attribution of causality outside of controlled-environment experiments continues to be a problem for ecological risk assessment, particularly at the level of complex systems, in spite of progress in understanding and defining the problem during the past 10 years. Several important papers have been written, all of which are well summarized by Suter (1993). The central problem is the so-called ecological fallacy, the assumption that measured differences between the environments inhabited by subject populations of interest are really the cause of differences in the frequency of some disease. The differences actually may be attributable to some unconsidered factor, and correlation does not prove causation. Koch's postulates have been proposed as a "standard" means of testing causality (Woodman and Cowling, 1987), but they are so specific for diseases as stressors that other authors have sought means for generalizing more broadly from them. One of these general treatments is that of Hill (1965), adapted by Suter (1993). Nine "factors" are suggested, a majority of which must be fulfilled:

- *Strength*—a steep exposure-response curve.
- *Consistency*—consistent association of an effect with a hypothesized cause.
- *Specificity*—the more specific the cause, the more convincing the association with an effect.
- *Temporality*—a cause must always precede its effects.
- *Biological gradient*—an effect should increase with increasing exposure.
- *Plausibility*—the underlying theory should make it plausible that the effect resulted from the cause.
- *Coherence*—the implicit relationships should be consistent with all available evidence.
- *Experiment*—the changes in effects should follow experimental treatments representing the hypothesized cause.
- *Analogy*—the cause-and-effect relationship should be similar to some well-known examples.

The point is that multiple lines of evidence usually are needed to ensure a reasonable inference of causality. Experimental proof is simple and ideal, but relatively few situations can be reduced to a controlled-experiment situation. Thus, there is a need to incorporate not only the uncertainty deriving from confidence intervals around data, when available, but also uncertainty as to what is or is not a causal relationship. Assembling the multiple lines of evidence for the causality of stressor effects at

single sites, or for larger systems such as lakes, will usually require evidence from other similar systems and modeling for the site under direct consideration. The aquatic food web model described previously (Vanni et al., 1992) is a good example. Field observations of fish, plankton, and nutrients might never have been seen as having the strength, consistency, specificity, coherence, and plausibility to attribute one or another as causal. Yet when the same results are viewed in the context of linked relationships in a simulation model, they can be seen as causal. Given the large body of pre-existing experimental work and theory for food chain coupling, a structured model incorporating a "cascade" of linkages is plausible and the need for a very large uncertainty term is reduced.

Two major questions remain in regard to the causality-uncertainty tradeoff: What is the range of other physical or biological conditions (e.g., in turbid reservoirs or river reaches) where one might now conclude these same effects to be causally linked to equivalent manipulations? And to what extent can causality be inferred in quite different systems where the underlying theory is less well developed or experimental field testing is only partially complete? Some answers to these questions are available from thorough retro-assessments of past decisions (sometimes viewed as mistakes) that provide unique opportunities to learn how risk assessments (often of chemical stressors) can be applied to complex systems. Although most risk assessments imply a decision-predictive situation, models built from retro-assessments go a long way to aid understanding of causality and to improving future applications.

6. ECOLOGICAL RESPONSE ANALYSIS: RELATING MEASUREMENT AND ASSESSMENT ENDPOINTS

In most ecological risk assessments, the measurement endpoints are not necessarily the same as the assessment endpoints. To characterize effects associated with stressor exposures, extrapolations must be made. These include extrapolations among taxa, from laboratory to field conditions, across the ecological hierarchy, and across spatial and temporal scales. The need for extrapolation in risk assessment is obvious; the problem is to assess the uncertainty in these extrapolations.

6.1. Extrapolations Among Different Taxa

An important issue to consider in the extrapolation of effects is the difference in sensitivity among species. Suter (1993) describes two types of taxonomic extrapolations. One involves extrapolating from the test data for one species (e.g., fathead minnow) to predict the response of a second species (e.g., rainbow trout). The second type of extrapolation is from data for one or a few species to predict the sensitivity distribution for the whole community exposed to the stressor.

Species-to-species extrapolations are based largely on regression analyses. Kenaga (1978, 1979) regressed LC_{50}s and LD_{50}s for all combinations for eight terrestrial and aquatic species using results from toxicity tests with 75 pesticides. Similar regression analyses have been performed for a variety of aquatic species and chemicals (e.g., Maki, 1979; Sloof et al., 1986). Regression between pairs of species allows one to predict responses for only the few species that have been frequently tested. Ecological risk assessment often requires assessment of effects on species that are not generally tested. Suter et al. (1983) devised an approach to address this shortcoming based on taxonomic relationships between the test species and species of interest. For aquatic organisms, regressions were performed, for example, on all pairs of species in a common genus and on all pairs of genera within a common family. Extrapolations are then made between taxa having the next higher taxonomic level in common. This approach is based on the assumption that differences in sensitivity for aquatic organisms increase with increasing taxonomic distance.

No similar analysis has been performed for terrestrial species, although several researchers have compared species sensitivities to chemicals. Peakall and Tucker (1985) compared rats with starlings, starlings with red-winged blackbirds, and mallard ducks with bullfrogs. These comparisons show that the ratio of LD_{50} values for the bird species fall largely within one order of magnitude, while the LD_{50} ratio values for starlings and rats, and for mallards and bullfrogs were more widely dispersed—generally between two and three orders of magnitude. The differences in sensitivity to

chemicals increased as the phylogenetic differences increased. This finding is similar to that of Suter et al. (1983) for aquatic species. These data also indicate that a moderate level of uncertainty is associated with extrapolations to closely related species, but a great deal of uncertainty exists with respect to extrapolations between distant taxa or for a wide variety of unrelated taxa.

A number of extrapolation approaches have been proposed to develop chemical-specific criteria to protect an aggregate of species that might be exposed in an ecosystem. The approaches fit a distribution to LC_{50}, NOEL, or other endpoints of single-species tests for the species of interest and estimate a "safe" concentration for a certain percentile of the distribution of species tested (Stephen et al., 1985; Kooijman, 1987; van Straalen and Denneman, 1989; Aldenberg and Slob, 1991; Wagner and Lokke, 1991). In concept, such a model would provide a concentration that would protect a specified percentage of the species (e.g., 90 percent) with a specified level of confidence. This type of extrapolation, which is used to develop EPA's ambient water quality criteria, is based on the assumption that the laboratory species for which data are available are representative of their field counterparts and that the sample of species is random. These assumptions are not likely to be met in most extrapolations since only a small percentage of species have been used in toxicity tests (e.g., approximately 6 percent of freshwater fish have been tested for at least one chemical) and most species are tested because of their sensitivity and economic importance.. These extrapolation methods were recently reviewed by OECD (1991) and Smith and Cairns (1993) and the inability of single-species tests to account for system-level effects was noted.

6.2. Extrapolations From Laboratory to Field Conditions

Laboratory tests measure stressor-response relationships under well-controlled conditions. Such tests may not mimic the environmental conditions, organism conditions, exposure conditions, or, more important, the biological interactions that occur in the field. Thus, several approaches have been used to facilitate extrapolations from laboratory effects data to predict effects in the field.

To address differences in environmental conditions between the laboratory and field for aquatic organisms, researchers have developed regressions for a few variables such as temperature and toxicity (Mayer and Ellersieck, 1986), pH and toxicity (Mayer and Ellersieck, 1986), and water hardness and toxicity (U.S. EPA, 1985). At present, no extrapolation models have been developed for addressing laboratory and field differences in concentrations of organic and inorganic nutrients, suspended sediments, total biomass, and other factors that are known to affect the availability of chemicals to aquatic biota. The magnitude of influence these factors exert on the concentration-

response relationship is unknown, but most laboratory test models are assumed to be conservative models; that is, the addition of sediments to the test system is expected to reduce bioavailability of the introduced chemical and its toxicity to test organisms.

Differences in sensitivity between wild and laboratory strains of test organisms have been evaluated for a few aquatic and terrestrial species. McEwen et al. (1973) found no differences in sensitivity to chemicals between pen-reared and wild-trapped pheasants. Peakall and Tucker (1985) reported that wild-caught *Daphnia magna* and *D. magna* from a long-held laboratory culture were equally sensitive to chemical exposures. In general, field experiments suggest that when environmental conditions are equivalent, organisms in the field respond at about the same exposure level as organisms in the laboratory (Crossland, 1982; Hansen and Garton, 1982).

Exposure conditions in the laboratory are generally as a single pulse, with concentrations diminishing with time. Field exposures are likely to be much more erratic. Although few attempts have been made to design laboratory tests to mimic exposures under field conditions, more complex exposure regimes could be incorporated into testing to improve confidence in this aspect of extrapolation. Aside from this, continuous exposure tests likely represent an upper bound of field exposure conditions. LC_{50} values for flow-through aquatic tests were found to be consistently lower than LC_{50} values from static tests for the same species, by ratios of 0.12 to 8.3 (Mayer and Ellersieck, 1986).

A more direct approach that should account for environmental and ecological factors is the use of mesocosms, artificial streams, or controlled field studies that more closely mimic field conditions. Boyle et al. (1985) used experimental pond systems to evaluate the effects of fluorene on the ecological structure and function of these aquatic systems and to evaluate the predictability of laboratory toxicity tests for them. They found that phytoplankton, zooplankton, and aquatic insect communities were much less sensitive in the pond systems than expected, based on laboratory test data. Taking this one step further, Sheehan et al. (1987) used field data on the toxicity of two insecticides, relative to their respective LC_{50} values from laboratory tests, to predict reductions in aquatic macroinvertebrate numbers and biomass for exposed pond ecosystems. These data were then used as "benchmarks" for extrapolations of chemicals with similar LC_{50} values. In situ aquatic enclosures also have been used to reduce uncertainty in extrapolations from test to natural systems by incorporating natural environmental factors and ecological functions in the test model (e.g., Kaushik et al., 1985).

6.3. Extrapolations Across Ecological Hierarchy or Organization Levels

Extrapolation of expected ecological response patterns from the species and population level to ecosystems is a new field of inquiry. The approach outlined above—i.e., aggregating from effects on individuals ("individual-based models"), to effects on populations, to effects on ecosystem processes (Cai et al., 1993)—is only one approach, with limited applicability. A comparative test of this simple extrapolation approach using No Observed Effect Concentrations (NOECs) for single species and ecosystems is reported by Sloof et al. (1986) and by Suter (1993), with satisfactory results. The problem may be best understood by giving the analogy that the effects of a stressor on the whole human body cannot be projected well simply from measurements of blood pressure on one arm; good circulation is a much more holistic concept.

A more sophisticated approach requires articulation of the unique properties of each level of organization, something like the "criterion" characteristics proposed by Allen and Hoekstra (1992). Following their (and other) proposals, researchers and assessors must recognize the unique properties that are afforded each level of organization—properties potentially at risk from effects expressed at a lower level of organization. For many reasons, effects on these "criterion" properties are not included in the simple "aggregating up" paradigm of Cai et al. (1993) or Sloof et al. (1986). A possible example is the property of "integrity" often associated with ecosystems. Aggregating the toxicity effects on earthworm individuals up to a cumulative effect in the form of decreased litter decomposition and slower nutrient cycling does not provide an estimate of the loss of system integrity at the higher level of organization. In the context of this discussion, system integrity is a holistic idea, as with human health. Although means for measuring it are improving gradually, a substantial body of new work is required if risks to the health of the higher levels of organization are to be addressed.

6.4. Extrapolations Across Spatial and Temporal Scales

As noted above, many problems have been encountered in extrapolation between species and between rates of exposure. With experience in extrapolations for human health risk assessments, some of these problems have been addressed. The corresponding problems in field environments, and with exposures that are cumulative over several decades, are just now beginning to be addressed. The discussion by Suter (1993) is illustrative. Citing Reckhow (1983), he suggests that for extrapolation between ecosystems (i.e., spatially), the assessor should first define an exposure-response model for a measure corresponding to the ecosystem-level assessment endpoint. The assessor also must define

and adopt tools (models) that describe the tested system and the systems to which the assessor wishes to apply the exposure-response relationship. Multivariate statistical tests could be used to estimate the probability that the study ecosystems to be evaluated belong to the same state space as the ecosystems used to develop the exposure-response model. If the assessor decides that the two sets of ecosystems are not the same, the model can be respecified so that it is applicable to the systems of interest and the assessor can proceed with the extrapolation. This approach has not yet been applied to effects assessment, however, and much further testing is needed.

The same questions have to be considered when risk assessments are applied to evaluate effects over large areas such as landscapes and regions. At this scale, the use of insecticides that may leach into streams or wetland drainage programs that can have large cumulative effects tend to alter a property of the landscape that goes unrecognized until it no longer functions. Important steps toward understanding and modeling system functions and dynamics at this scale is reflected in the work of Wu and Vankat (1991).

This still leaves the problem of extrapolation between temporal scales (e.g., the estimation of risk to ecosystem endpoints after 40 to 80 years of exposure to lead, or to acidic deposition, based on a few 4-year experiments or a retro-assessment). Such extrapolations have been projected in the course of acid-deposition assessments with some confidence for physicochemical endpoints in the soil, but with very little confidence in projections of long-term biological responses. The reason, of course, is our limited understanding of the mechanisms of aging of long-lived species making up ecosystems or of how aging processes are affected by stressors. In the absence of some reasonable idea of the mechanisms that would be operating over the long term, or even the mechanisms that are operating in the short term, extrapolations across long time scales have little foundation. Given the improvements now emerging in long-term global circulation models (GCMs) and in forest succession models, the situation could change.

7. STRESS REGIME-RESPONSE PROFILES

Results from the characterization of ecological effects should be summarized in a stress regime-response profile, which is a synthesis of the effects—both direct and indirect, short term to long term—that are identified in a characterization analysis. The profile relates the magnitude, duration, frequency, and timing of effects quantitatively to the magnitude and duration of exposures. To be informative, an expression of the stress regime-response profile should include the shape of the relationship (usually some curvelinear form, such as a dose-response relationship), upper and lower limits, and uncertainty bands. For primary effects, many elements of a stress regime-response profile may be shown schematically, such as the concentration-response or dose-response curves and uncertainty bands for chemical agents. The stress regime-response profile also could be displayed as a probability distribution, at least for clearly understood relationships.

Displaying graphically or quantitatively the relationship between the primary agent (and its stress regime) and secondary or indirect effects on populations or ecosystems is more difficult. In these situations, the profile should consist of a written synthesis of the linkages between the stressor and observed responses in the system. The relationships may be oblique, in some cases, and the characterization requires description of a large number of interactions intermediate between the cause and the secondary effects.

7.1. Stress Regime-Response Relationships for Chemicals or Physical Stressors Acting on Individuals and Populations, Short- to Mid-Term

Stress regime-response profile diagrams are frequently produced for chemical toxicity tests but are rarely documented for physical stressors. Examples of chemical concentration-response profiles are presented in figures 5-4 and 5-5, showing a clear relationship between chemical concentration, exposure duration, and organism response. The response surface shown in figure 5-5 integrates exposure duration and concentration into the response profile. These profiles do not show the uncertainty in the concentration-response data; however, the uncertainty band could be added easily to enhance the value of the profile. Barnthouse et al. (1987) provide an example of a concentration-response profile with uncertainty bands that provide a clear characterization of effects and extrapolation of uncertainties.

The synthesis models described above could be applied to more complex systems. Barnthouse (1993) provides an example of a profile that integrates exposure-response data and uncertainties for individuals and populations such as might be found in a stressed ecosystem (figure 5-7). Such a

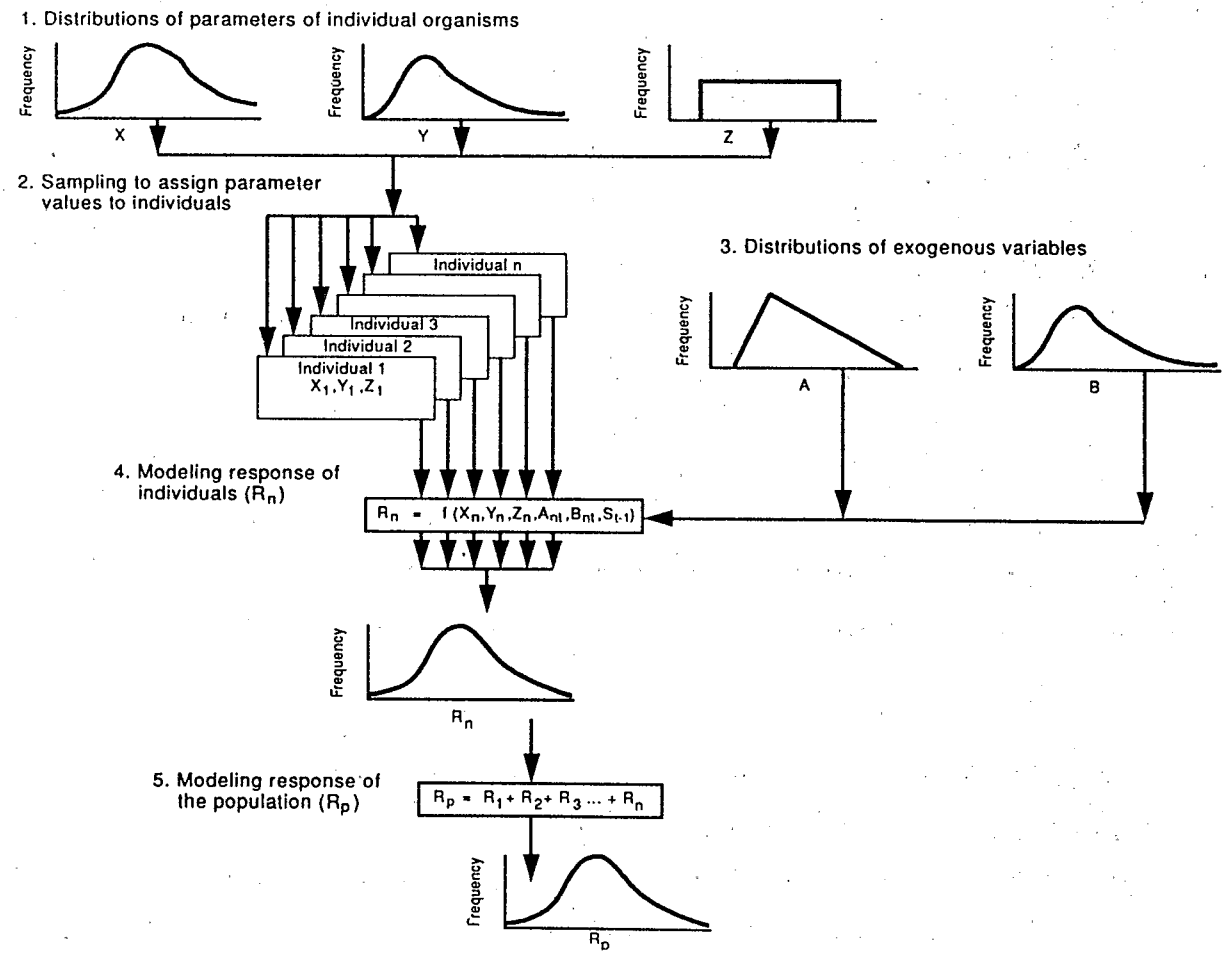

Figure 5-7. A schematic representation of individual-based population modeling X_n, Y_n, and Z_n showing characteristics of an individual organism *n* such as size and leaf area. A_{nt} and B_{nt} are characteristics of the environment experienced by individual *n* at time *t* such as temperature, pollutant concentrations, and prey availability. S_{t-1} is the state of the organism at the previous time step. R_n is response of individuals such as death or maturation. R_p is response of the population such as abundance or harvestable biomass (from Barnthouse, 1993).

profile could be expanded to be more comprehensive by adding dimensions to account for a larger number of populations of interest in an exposed system. (This profile model also could be extended to describe ecosystem-level effects—in addition to individual and population-level effects—if the ecosystem-level stress regime-response relationships are understood and the requisite ecosystem data are gathered.)

The multiple dimensions of a stressor regime-response profile, in relation to the natural variability of measurement or assessment endpoints, can be understood in terms of the "assessment hypervolume" proposed by Landis (in press), and illustrated in figure 5-8. A 95 percent confidence region in multidimensional system space is shown as a box within which the historical trajectory of a measurement endpoint can be seen as a response over time. An alternate response—one that carries the endpoint trajectory outside the variability of the defined hypervolume—would be a significant alteration of the system if it were a stressor-induced response. Its expression here as a trajectory over time conveys our need to understand a response profile as a departure beyond the known variability of the measurement endpoint. Also, the response may not be immediate or even linear over time, depending on the dynamics of natural factors or interactions over the timeframe involved.

7.2. Stressor-Response Relationships Attributable to Chemicals and Physical or Habitat Alterations Acting on Ecosystem Function and Stability Characteristics, for Large Areas, Long Term

In principle, the stressor-response profiles described above for stressors acting on individuals and populations seem to constitute a reasonable concept for the same stressors when acting on ecosystems. For the reasons discussed in earlier sections, a simple "aggregating up" of effects expressed at the population level can lead to a first approximation of effects on certain ecosystem functions, at least locally. Where this is the case, essentially the same response profile could be applied to specific ecosystem functions, changing only the dependent variable.

Although the general stress regime-response model proposed in the previous section for individuals and populations is theoretically applicable for characterizing structural and functional effects on ecosystems, such profiles have not yet been generated. At present, we are able to describe concentration/dose-response relationships for only a few illustrative chemicals and for only a few of the important ecosystem functions. Moreover, it is unclear whether such profiles can be generalized to other key functions such as multispecies feeding, net production, and net migration (e.g., for birds). Similar uncertainties exist for characterizing physical stressors and their effects on ecosystem

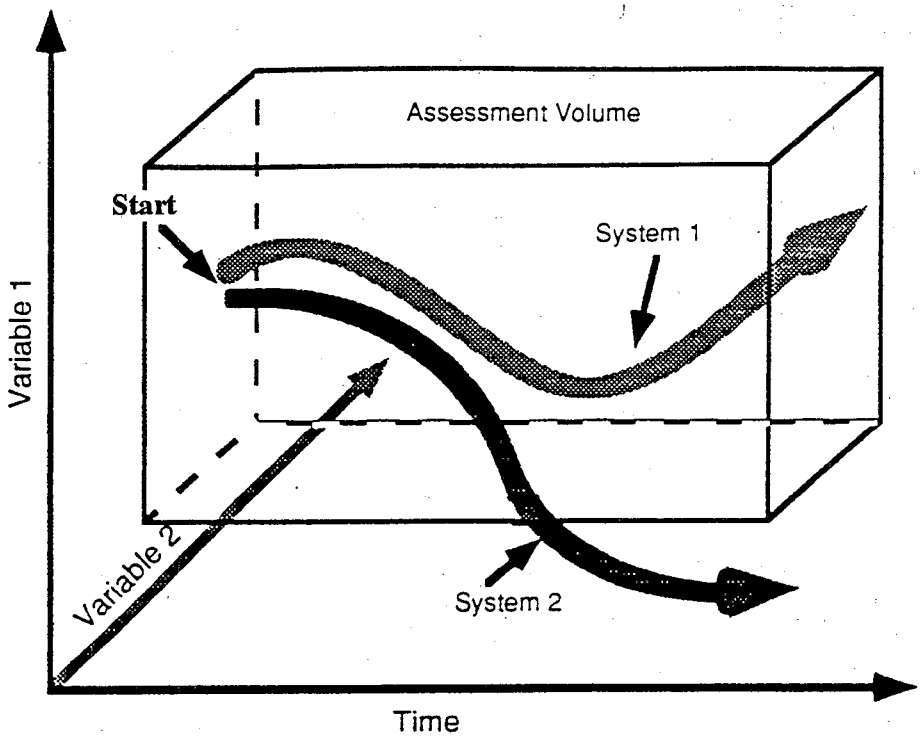

Figure 5-8. **An endpoint assessment "hypervolume" reflecting the normal variance of an endpoint measure, but also the possibility (a response profile) that under a new stressor regime, the endpoint has a certain probability of leaving the normal variance of the assessment volume over some period of time (after Landis et al., in press).**

function, partly because metrics of these stressors that can be satisfactorily aggregated or averaged are not available.

Although we probably know enough about what is needed in a response profile for ecosystem function to begin asking the right questions or adapting certain models to the problem, it is premature to anticipate how difficult such a synthesis will be. Two questions that need to be considered are: How essential is this part of the puzzle in comparison with other gaps in our knowledge noted earlier? And, now that we see an array of needs, is there a way in which a wide variety of them can be met through one coherent program of experiments and models, as is being done in the global change research program?

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Issue Paper
on

BIOLOGICAL STRESSORS

Daniel Simberloff
Department of Biological Science
Florida State University
Tallahassee, Florida

Martin Alexander
Department of Soil, Crop, and Atmospheric Sciences
Cornell University
Ithaca, New York

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U.S. Environmental Protection Agency

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1. INTRODUCTION AND SCOPE

Several factors must be considered in assessing the risk posed by a biological stressor. The stressor must first invade a particular ecosystem. Yet even if it survives, it need not have a discernible impact on that ecosystem. The probability of either successful invasion or disruptive effect is highly uncertain. This uncertainty is the basis for some paranoia concerning the outcome of the release of novel organisms, particularly genetically engineered organisms, into managed or natural ecosystems. Conversely, despite the uncertainty, some biologists are unconcerned about novel introductions. Both groups muster evidence to support their contentions, but rarely have members of either camp attempted a formal risk assessment for such stressors. That some introduced organisms have become established and have caused major disruptions proves that a risk exists, especially for organisms about which there is little information—that is, most organisms. On the other hand, the likely failure of many—and probably most—introduced species to become established or to have a detrimental impact on an ecosystem (Simberloff, 1981) suggests that the risk is small for most species. Data do not exist, however, to assess how small is small or to make a determination for most potential biological stressors about the likelihood of being problematic. Further, some of those relatively few introduced species that have survived have generated huge costs, leading to the enormous concern about biological stressors.

Thus the risk assessor concerned with biological stressors probably will encounter two disparate groups of experts. The first will emphasize the evidence for problems and will cite acknowledged instances of major ecological impact. The other will downplay such evidence as idiosyncratic and will present plausible arguments that the average introduced species will be very unlikely to create an environmental stress. Adherents of the first camp would respond that the effect of the average introduced species is irrelevant; one should be interested not in the mean effect but in that of the outliers. This expert background noise characterizes the setting for risk assessment of biological stressors.

We begin in section 2 by outlining the key differences between biological and other types of stressors, including differences that produce the uncertainty mentioned above. The main differences are that (1) biological stressors reproduce and multiply, (2) they disperse by myriad means, often in large "jumps" that are inherently very difficult to predict, (3) their interactions with other living entities are extremely difficult to predict, and (4) they evolve, and this evolution has large elements of chance.

The risk presented by a biological stressor rests on four factors: survival, multiplication, dispersal, and ecological effects. In section 3 we discuss what general information is available on survival, proliferation, and dispersal, with an emphasis on what is not known generically about these processes. Then, in section 4 we consider the myriad kinds of effects that biological stressors can have on populations, communities, and ecosystems. In sections 5 and 6 we discuss how the uncertainties associated with biological stressors, and various characteristics of biological as opposed to chemical and physical stressors, complicate the choice of assessment endpoints. In section 7 we consider ways in which recovery from biological stressors resembles and also differs from recovery from chemical and physical stressors.

To a large extent, the effects at the community and ecosystem levels that one might expect of biological stressors, and therefore the risks one sees as posed by them, are conditioned by one's conception of communities and of ecosystems. There is a spectrum of opinion about the nature of communities and their importance as a level of biological organization. The *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) represents one extreme of this spectrum: "Community—An assemblage of populations of different species within a specified location in space and time." The other extreme is the superorganismic community conceived by Clements (1936), in which the community has an integrity and organic nature quite comparable to that of an individual, with the different populations interacting in stylized ways and with various functions, such as energy flow, fully analogous to the physiological functions of individuals.

The controversy over the nature of communities is a very old one (Simberloff, 1980; Underwood, 1986), remains a burning issue among ecologists, and is not be resolved in this paper. Suffice it to say that both ends of the spectrum are still represented. Thus Wilson and Sober (1989) write about holistic communities and ecosystems in a very Clementsian vein, while the definition in the *Framework Report*, though extreme, is held by some current ecologists. Ecologists hold a similar range of views on ecosystems, with some perceiving ecosystem processes like nutrient cycling as analogs to the physiology of organisms (Rambler et al., 1988) and others viewing them as the inevitable and uninteresting by-product of the co-occurrence of multiple populations in a particular physical setting. The debate about emergent properties of communities and ecosystems (Simberloff, 1980; Wilson and Sober, 1989) is simply another aspect of this long-standing controversy: Can every aspect of the structure and function of communities and ecosystems be comprehended by a full understanding of their component populations and the physical setting, or do certain features emerge

only by virtue of the complex interactions among the populations, so that these features are intractable to a study by reductionist methods?

Clearly, if one sees the community or ecosystem as simply an epiphenomenal consequence of the co-occurrence of populations, one is not likely to be concerned with the effects of stressors on traits at these levels. Thus assessment endpoints are unlikely to be community or ecosystem features, although such features might be measurement endpoints for individual or population traits of interest. Rather, the effects of a biological stressor are likely to be considered, and sought, as the sum of its effects on the various component populations. If, on the other hand, one considers the community or ecosystem to be a major level of organization, comprising holistic entities that might even have emergent properties, one is likely to seek effects of stressors on traits at this level, particularly emergent ones. One might well expect a new biological entity to sunder completely the previous community and, through its interactions with other populations and with the physical environment, to generate an entirely new community. That is, a small change in the species composition could lead to a very large change in the functioning of the community and ecosystem. In sum, one would seek very different effects and select very different endpoints, depending on one's view of communities and ecosystems.

In this paper, we have chosen not to restrict ourselves to the *Framework Report*'s definition of "community" and to consider effects that span the spectrum from effects on individuals and populations through effects on communities and ecosystems. In our view, regardless of whether communities and ecosystems are holistic, they have features that can be measured only at that level, and even if they are only compound properties (e.g., various diversity measures) rather than emergent ones, the public is concerned with some of these features, and laws even address them. Further, it is entirely possible that there are great differences among communities and ecosystems, with some highly organized and others simply collections of coexisting populations.

In this chapter we conceive of biological stressors as living entities added to an ecosystem. We do not address other sorts of stress generated by changes in the living organisms of a system, even though such stresses may be extremely important. For example, the loss of species to human harvest can have drastic consequences. Consider the 100-fold decline of the eastern oyster (*Crassostrea virginica*), which has had myriad effects. At the turn of the century, oysters removed particles of 2 to 20 microns in size from Chesapeake Bay by filtering the entire volume of the bay's water approximately weekly. Today oysters are so scarce that the bay water is filtered only once a year (Newell, 1988). Because suspended particles can be organisms themselves and are crucial in

transporting nutrients and are a factor in light penetration and sedimentation, such drastic alteration in their cycling must have substantially affected the entire ecosystem. Similarly, the loss of various bird species to hunting must have affected plant dispersal secondarily in many ecosystems, with subsequent effects on species composition and other ecosystem features. We view the stressor in these instances, however, as the human activity that produced the species decline, not the species undergoing the decline.

We consider briefly the assessment of risks posed by particular pathways or commodities that need not be biological stressors themselves but may carry large numbers of potential biological stressors. For example, ballast water used in international shipping carries living organisms of many species all over the globe (Carlton and Geller, 1993). Both living commodities (e.g., ornamental plants, fish farm stock) and nonliving commodities (e.g., unprocessed logs) may be agents of inadvertent movement of biological stressors.

Finally, we consider biological stressors to include any species introduced to regions not contiguous with its original geographic range, thus encompassing organisms moved between regions of the continental United States. A recent report published by the U.S. Congress' Office of Technology Assessment (OTA), *Harmful Non-indigenous Species in the United States* (1993), has numerous examples of the effects we discuss in section 4, plus an extended discussion of risk assessment of biological stressors. The report generally takes the opposite tack, however, considering the impacts only of species originating outside the continental United States. We see no logical or scientific justification for this narrowed scope. For example, the rainbow trout, though indigenous to the continental United States, has been introduced widely beyond its natural range to the detriment of brook trout and other native species, as noted in the OTA report. Even intrastate movement can be problematic. The yellow bush lupine (*Lupinus arboreus*), introduced from San Francisco, has greatly modified the entire dune community at Humboldt Bay by its survival in the nutrient-poor dune mat and subsequent modification of many features of the environment (references in Simberloff, 1991). In any event, it would be perverse to exclude intracontinental introductions from our purview in light of the great concern about the movement of organisms of non-U.S. origin even after they establish an American beachhead. For example, the OTA report assesses the spread and risks posed in the United States by such Eurasian scourges as the zebra mussel and purple loosestrife, rather than simply tallying them as established and thus not worthy of further attention.

2. KEY DIFFERENCES BETWEEN BIOLOGICAL STRESSORS AND CHEMICAL AND PHYSICAL STRESSORS

The *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) has many facets that would be useful in assessing risks imposed by a biological stressor such as an introduced species or a genetically engineered organism. For example, the *Framework Report* separates exposure and hazard analysis, and the same separation is appropriate for biological agents. Similarly, the emphasis in the *Framework Report* on the scope of the assessment and the breadth of possible risks to be considered applies equally to biological stressors. Certain characteristics of living organisms, however, vitiate the utility of a framework written primarily with physical and chemical stressors in mind.

First, all living organisms reproduce and, in the course of reproduction, may multiply. With a chemical stressor, simply ceasing to impose the stress inevitably leads to a lessening of the risk of any ecological effect as the molecule breaks down naturally or as various mitigation procedures facilitate the breakdown of the molecule and/or its breakdown products or their transport from the site. With a living organism, arresting its introduction to a site need not substantially lower the risk of an ecological effect—even if the organism dies—so long as it can reproduce there.

For some introduced species, a population increases only slightly initially, then remains stable for many generations as individuals just replace themselves. In 1870, about 12 pairs of the Old World tree sparrow (*Passer montanus*) were released in St. Louis, Missouri. They quickly established a persistent population, estimated at 2,500 to 25,000 individuals, which has been restricted for over a century to St. Louis and adjacent areas of Missouri and Illinois (Lever, 1987). Stragglers are occasionally reported as far away as Wisconsin, but the population never expands in range or size. The Mediterranean fruit fly (*Ceratitis capitata*), a major pest associated with fruit and vegetables, appears to have established a persistent but generally small population in southern California despite repeated, extensive eradication campaigns carried out by the U.S. Department of Agriculture (USDA). A blue-ribbon committee established by the University of California (Berkeley) concluded that the sporadic outbreaks in the Los Angeles area are not likely the result of repeated introductions, but manifestations of a sparse population that has persisted for at least 5 years and possibly much longer (Van Steenwyk, 1990; Carey, 1991). For microbial risk assessment, the situation may be considerably worse than these metazoan examples indicate. Microorganisms can persist for long periods, by various means (see section 3.1), at such low numbers as to be undetectable, yet can, under certain circumstances, grow rapidly and spread. Chemical and physical stressors normally

would not have such abilities, though a sequestered chemical could be liberated by some physical or chemical perturbation.

Reproduction and low-level, long-term maintenance at one site, however, are probably not the most common trajectory of a surviving introduced species. Most introduced species that survive at all probably increase in numbers and spread, at least initially. Certainly both the tree sparrow and the Mediterranean fruit fly increased, although the initial increase was limited and there was almost no geographic spread. A more likely result, if an introduced species survives, is substantial population increase associated with at least some geographic spread. Although rate of increase is limited by such life-history characteristics as generation length and fecundity, all species have the capacity for exponential increase in an appropriate environment.

From the standpoint of potential ecological effects, the possibility of geographic spread is of even greater concern than the likelihood of local increase. Indeed, all species have means of dispersal. Because the nature of this dispersal process and of range changes generally has been extensively studied (e.g., Johnson, 1969; Pielou, 1979), some assessment of risk of spread can be attempted. Yet this process of spread, particularly for species that move of their own volition rather than through passive means (e.g., by wind, water currents, or phoresy), is qualitatively different from that of chemical or physical stressors. Indeed, many living organisms are capable of dispersal over extremely long distances, and such dispersal is likely to make risk assessment particularly difficult, a problem discussed in more detail in section 3.3. It is also worth noting that even species that depend on either physical processes or other species for transport can increase their ranges substantially through increments because they can reproduce and multiply as each generation reaches a new suitable site.

For example, although the Asian fungus-causing Dutch elm disease (*Ophiostoma ulmi*) requires two bark beetles for dispersal in North America, the fungus spread rapidly from its initial site of introduction in the early 20th century to infect most elms throughout the East (Elton, 1958; von Broembsen, 1989). Interestingly, one of its beetle vectors, introduced from Europe at approximately the same time as the fungus itself, spread more rapidly than the fungus (Elton, 1958); nevertheless, the fungus ultimately reached almost the entire range of its host.

Another characteristic difference between some biological stressors and chemical or physical ones is the range of stressed organisms. For example, pathogens affecting animals, plants, and microorganisms can be particularly host specific, as can parasitoids of insects and some phytophagous insects. Such limited ranges of susceptible species are not characteristic of chemical or physical

stressors, which typically act on a wide range of species. Some biological stressors similarly affect a large number of species (see section 4.1).

A key difference between chemical or physical stressors and biological ones is that the latter can evolve, and this evolution can either increase or decrease the risk of ecological effects. For example, the Dutch elm disease fungus was introduced to North America from Europe on infected logs. In North America, it apparently evolved more pathogenic strains, which have been implicated in a recent new outbreak of the disease in Europe (von Broembsen, 1989). On the other hand, numerous introduced pathogens and their hosts have coevolved such that initially virulent diseases have become relatively benign (Ewald, 1983). For the myxoma virus introduced to control rabbits in Australia, the virus has become attenuated and the rabbits resistant (Krebs, 1985; Williamson, 1992). Yet evolution itself has numerous unpredictable aspects. For example, it may be severely constrained by which mutations happen to arise during a particular period or by which chromosomal crossovers occur. Such aleatory components of evolution vastly complicate risk assessment and differ qualitatively from any process associated with chemical or physical stressors.

Another way in which biological stressors characteristically differ from chemical and physical stressors is that, once established, they are either impossible or extremely difficult to remove. Extremely expensive and controversial attempts to eradicate even small insect pest populations have rarely been successful (Dahlsten, 1986). Eradication of the coypu in Great Britain (see section 7) stands out among many failures to eliminate introduced pest vertebrates.

A biological stressor may generate another kind of bizarre problem, a development completely different from anything encountered with chemical and physical stressors—the stressor may become a valued ecosystem component. For example, although eucalyptus introduced to Angel Island in San Francisco Bay provide a much less suitable habitat than native plants for the resident native animal populations, a California State Department of Parks and Recreation plan to remove them generated a firestorm of protest from people who found the plant species aesthetically pleasing (Azevedo, 1990). Similarly, the population of mountain goats (*Oreamnos americanus*) introduced by hunters to the mountains of Washington state's Olympic Peninsula in the 1920s has grown and spread. Now the goats are destroying native plants, including rare species, in the Olympic National Park. A National Park Service plan to eliminate almost all of them, however, alarmed not only animal rights groups but hunters (Luoma, 1989). This scenario is played out repeatedly with game mammals introduced by hunters that destroy native plants; feral pigs are of particular concern. The general problem has been

exacerbated recently with the advent of animal rights groups that do not oppose the removal of the animals per se but object to killing them by any currently practical means.

3. SURVIVAL, PROLIFERATION, AND DISPERSAL

For a biological stressor present at a particular site, four factors must be considered in a risk assessment: (1) probability that the organism will survive; (2) probability that the survivor will multiply to yield a larger population; (3) probability that the species will disperse from the initial site to another location at which establishment is possible; and (4) probability that the stressor will be harmful. If probabilities for the first and fourth factors are zero (i.e., the species does not survive or has no harmful ecological effects), the probability of a deleterious environmental impact is zero. If the probability for the third is zero (i.e., the organism fails to spread), the probability of environmental stress resulting other than at the initial site need not be zero. One must also consider the probability of genetic information associated with environmental impact being transferred to another species; this possibility is discussed in section 4.1. If such a "lateral transfer" occurs, a similar analysis of survival, multiplication, dispersal, and harmful effects must be conducted for the organism receiving the genetic information (Alexander, 1985a). A similar approach to risk analysis is used by USDA (Orr et al., 1993).

3.1. Survival

The number of propagules of a released biological stressor is likely to be critical to its initial reproduction and multiplication and thus to the risk of a potential ecological effect. Every species has a "minimum viable population size" (Shaffer, 1981; Simberloff, 1988)—or better, range of population sizes—such that, when the population falls below this range, rapid extinction is likely because of a number of forces, all of which affect small populations more heavily than large ones. For example, some dioecious species experience difficulty finding mates at low densities, while other species may require group stimulation of ovarian development or mating (Simberloff, 1986a, 1988). These problems concerning breeding and increase at low population size, collectively termed the Allee effect, have been the target of some modeling efforts (Dennis, 1989). For a number of classes of introduced species, an increase in the number of propagules contributes to an increased probability of survival, although many other factors also contribute (Smallwood, 1990). Pimm (1991), studying game bird introductions with different numbers of propagules, found probability of survival to increase with the number of propagules and even claimed to find a sigmoidal shape for the curve.

Survival and initial increase also depend critically on the environment at the site of introduction. The percentage of surviving propagules varies with the particular organism and the environment (Liang et al., 1982). Some species can survive in one environment for many years with no

replication but disappear rapidly in other environments. The critical environmental differences may be quite subtle, as in the case of microorganisms that persist in one soil but decline to undetectable levels in another. Nonetheless, every species has a range of habitats compatible with survival and reproduction; often the range for survival is wider than it is for reproduction. For example, many plant species' geographic ranges are limited not by the ability of adults to survive, but by their ability to produce seed at the margins of their range. Thus individual plants that appear perfectly healthy at the margins may all derive from seeds produced in more central populations and be unable to produce seed that will expand the population further (e.g., Neilson and Wullstein, 1983). In fact, many species may exist as metapopulations (i.e., loosely connected sets of populations) in which a few "source" sites with ideal habitat produce dispersing individuals that colonize lower-quality "sinks" (Pulliam, 1988; Pulliam and Danielson, 1991). Whereas such a species would not be capable of maintaining itself in the sinks, these may constitute the majority of the range.

The vast majority of propagules of plants, animals, and microorganisms almost certainly die without issue because they end up in inappropriate habitats (e.g., terrestrial seeds land in water, microorganisms colonize too acidic or alkaline an environment, parasitic organisms fail to find a suitable host). Determining what exactly constitutes a receptive environment for a species often is not a trivial matter, but instead one that requires intensive experimental and natural historic research. Information on the survival of microorganisms comes chiefly from the public health and agricultural literature. Abundant data exist, but only for a few organisms that are important in diseases of humans, livestock, and agricultural crops, for a few bacteria used as indicators of fecal contamination, and for several bacteria of agronomic importance (Alexander, 1985a). This information shows clearly that certain microorganisms are able to persist, often for long periods, in environments in which they are not indigenous (Liang et al., 1982). Although the relative frequency among microorganisms of species that survive in alien habitats is unknown, we know that the rate of death of such introduced microorganisms is affected by a number of physical and chemical factors, including drying, moisture level, pH, solar radiation, salinity, and the presence of organic and inorganic toxins.

Moreover, although the survival of many microorganisms is not greatly affected by these stresses, they do not persist. For most potential microbial invaders, it is not the physical and chemical properties of the environments (at least of soils and waters) that preclude establishment. In laboratory tests, samples of soil or water that are sterilized (to render them free of other microorganisms) are readily colonized by species that do not—as well as those that do—have the capacity to survive or to proliferate in nonsterile samples of the same soil or water. Undoubtedly, the

activities of some of the resident populations constitute the basis of this biotic resistance to the successful establishment of an invader. The activities of concern may be competition for limited resources between the recent arrival and indigenous populations, parasitism, or grazing by protozoa or other predators. Although grazing pressure and the degree of competition and parasitism clearly vary by habitat, it is usually impossible to predict which community or habitat will be suitable for invasion, except for environments that have such adverse conditions because of physical or chemical properties that only species tolerant of those factors will be able to flourish.

Thus microbial species are precluded from establishment because they fail to compete with the indigenous microflora and as a result die from starvation or because they are susceptible to predation by protozoa and possibly metazoa. As in metazoans and plants, however, competition between introduced and native species is extremely difficult to document in the field (see section 4.1). Although microorganisms have parasites, parasitism is not known to eliminate introduced microorganisms. Nonetheless, parasites of microorganisms may be important when a species achieves substantial population size or biomass; an impact of parasites on a low-density population of a particular host species is unlikely. Again, the survival of a microorganism in one environment but not in another often is probably the result of differences in the competitive and/or predatory regimes. Data to support this assertion are sparse, however, and most studies are not very convincing. Moreover, the view that nutritionally fastidious microorganisms will not persist in natural environments runs counter to the facts; complex nutrition may affect the capacity of a species to proliferate, but it does not necessarily deter its survival.

To survive, an introduced microorganism must tolerate abiotic stresses. It must also avoid, evade, or cope with many competing microorganisms, the grazing activity of predators, and possibly, attack by parasites. Starvation is a major stress, and a survivor must be able to endure periods of nutrient deprivation. Some microorganisms overcome these obstacles by forming resting structures (i.e., the endospores of certain bacterial genera; cysts of many protozoa; and sclerotia, chlamydospores, conidia, and other structures of fungi). These resting structures can persist for many years. Even some bacteria and fungi with no such specialized structures will persist for months, years, and even decades. Data also suggest that certain bacteria enter a dormant state in which they appear to be injured in some manner. Injured bacteria that have been found in natural waters (Roszak and Colwell, 1987) and have undergone some physiological stress such that they will not multiply in the usual media used for their enumeration can be recovered in appropriately supplemented media. These "injured" bacteria generally would not be detected if one used conventional procedures; yet

they may endure for long periods and ultimately give rise to a population that grows and has a deleterious impact. In many instances, the number of surviving microorganisms may be below the level of detection and be deemed to be absent. Given suitable conditions, however, the few survivors will multiply and give rise to large and possibly harmful populations. A potentially major impact from a stressor initially below detection limits, however, is not characteristic of abiotic stressors.

Some metazoans and plants also have resting stages in which they can remain viable for long periods, even in a stressful environment. The seeds of many species may remain dormant buried in the soil for many years before germination—this is a typical trait of annuals and other short-lived species. Thus one cannot assume that a plant has disappeared from an area simply because no seedlings are observed. It is possible to assay soil for viable seeds but the procedure can be onerous, especially if seed density is low. Among animals, resistant eggs of nematodes, fairy shrimp, rotifers, mosquitoes, and other species correspond to the resting stages of microorganisms and seeds of plants. Similarly, sponge gemmules, bryozoan statocysts, and other life-history stages constitute persistent resting stages and often are characteristically produced when the environment becomes harsh.

Therefore, considerable uncertainty exists in predicting the probability of survival of microbiological, plant, and metazoan stressors, except for the very few groups that have been intensively studied. Methods exist for assessing survival, but in view of the frequent lack of understanding of the reasons for elimination or endurance and of the contribution of abiotic and biotic factors to death, only a few generalizations are possible.

3.2. Proliferation

For microorganisms, various terms are used to describe an increase in number or biomass. Although the term "growth" is often used for bacteria and other unicellular microorganisms, such growth actually refers to multiplication (i.e., increases in cell number or population size). Filamentous microorganisms (i.e., fungi, many algae, and some bacteria) may increase in biomass without a concomitant increase in countable numbers; thus, they grow without necessarily multiplying. Some specialists refer to this increase in number or biomass as colonization or, less commonly, establishment. To avoid semantic difficulties in this discussion, growth, multiplication, and colonization are all considered proliferation. For plants and metazoans, "population growth" and "multiplication" both mean an increase in number, while "colonization" refers to the establishment of propagules as well as a subsequent increase in number.

After an introduction, a propagule that does not diapause by some means can multiply rapidly. For an unlimited environment, numerous models of local population growth predict the initial trajectory (Krebs, 1985). For plants and metazoans, depending on the level of knowledge available, these models can incorporate such features as age and sex structure. If interactions with other species do not intervene, this local population growth can often be modeled quite accurately. Except for highly localized environmental effects, however, the main stresses imposed by an introduced species would be experienced at a much broader geographic scale as the initial population spreads to form a metapopulation or completely separate populations. Metapopulation dynamics are just beginning to be modeled (e.g., Hanski and Gilpin, 1991) and few empirical data are available to test the models. Moreover, the simple trajectories of single-species population growth are greatly complicated when interactions come into play. This is not to say that prediction is not possible in such instances, but that many data are needed and detailed observation and, often, experimentation are necessary to establish which interactions are crucial in limiting a population's growth.

The fact that proliferation of microorganisms occurs is obvious. It is attested to by the outbreak and spread of diseases of humans, other animals, and plants as well as the development of phytoplankton blooms, the spoilage of foods, and the appearance of large bacterial numbers on early emerging roots. The issue in risk analysis is not whether proliferation of any microorganism will take place, but whether the population density or biomass of a particular species will increase in a given environment. Currently the data bearing on this issue are not numerous.

Proliferation is essential for any species to have an environmental effect because the number of initial propagules is nearly always too small to be of ecological concern. Proliferation requires that nutrients that can be used by the particular organism be available. For most microorganisms, the limiting nutrient is carbon (C) because the supply of inorganic nutrients is generally not limiting. In those instances in which much readily degradable organic matter is present and that organic matter has a high C:N or C:P ratio, the limiting nutrient may be nitrogen (N) or phosphorus (P). For algae and photosynthetic bacteria, the limiting nutrient is inorganic—usually N or P. The episodic increase in the supply of limiting nutrients, however, is not sufficient to result in an increase in the abundance of an individual species requiring that nutrient. Many coexisting species may use the same nutrients. Which of these many species are able to respond and proliferate cannot generally be predicted. The sole exception is the environment in which a truly unique nutrient exists, and, apart from a few organic materials that support a limited range of microorganisms, the truly unique nutrients are host organisms. The host for a parasite constitutes a unique nutrient for a pathogen, although the

uniqueness requires that the parasite overcome the many barriers to infection (e.g., skin, cutin, phagocytes, lignified tissues, antibodies).

From the viewpoint of exposure analysis, for which microbial proliferation is an essential component, the magnitude of the increase in population size or biomass can be enormous. A few bacterial cells may multiply to yield populations of 10^6, 10^9, 10^{12}, or more cells, and the exposure rises in parallel with this increase in organism abundance. Similarly, a biomass of less than 1.0 ng (10^{-9} g) may increase to yield a biomass of 1.0 kg or more, such as can occur in algal blooms in aquatic environments. In this fashion, the ultimate exposure in the worst case is the maximum population size or biomass to which susceptible populations or communities are exposed.

Microbial rates of proliferation vary enormously (Alexander, 1985b). At one extreme are many soil bacteria; because of the slow turnover of organic material in soil not receiving recent additions of plant residues or leaf litter, no more than a few cell divisions may occur each year. At the other extreme are bacteria, fungi, algae, or protozoa that develop at very high rates and for which a doubling in cell numbers or biomass may require less than an hour. Unrestricted proliferation of a single cell with a doubling time of 1 hour would yield a population 8 million times larger after one day, a phenomenal increase in exposure. A key word is "unrestricted," because rarely do conditions permit unrestricted growth, just as in plants and metazoans discussed above. Yet few of the restrictions on microbial proliferation are characterized, other than nutrient limitations and host responses. Even these two have qualifications, moreover, since an environment in which microorganisms are limited by a particular organic nutrient often receives episodic inputs of that limiting nutrient. Similarly, a host species that typically exhibits resistance to parasites contains compromised or genetically more susceptible individuals, or it undergoes modifications because of environmental changes that permit rapid proliferation of a parasitic microorganism.

Maximum plant and metazoan reproductive rates also span an enormous range; generally the rate is inversely correlated with body size (Bonner, 1965; Fenchel, 1974). Reproductive rates are useful in assessing risk from a new species, but other factors often overweigh them. If a biological stressor is introduced to an ecosystem during or soon after a disturbance that greatly reduces potentially competing existing populations, a high reproductive rate may increase the probability of initial establishment and even dominance. Weedy plants with high reproductive rates often dominate locally after a hurricane or trail-clearing in a forest. Maximum rates measured under ideal conditions, however, probably bear little relation to realized rates under almost any field conditions, except perhaps at the outset of an introduction. Also, many species that are both common in nature and

ecologically important have low maximum reproductive rates, while many species with high reproductive potential are uncommon except in sparsely distributed disturbed areas. Thus the stress imposed on a system by a quickly reproducing organism may be temporary.

Studies of individual microorganisms in the laboratory provide hints of the traits that may result in rapid proliferation in nature, but no more than hints. In the absence of validation in nature, or at least in microcosms, they should be taken only as highly tentative suggestions. One might assume that a species with the fastest multiplication rate would be the one that proliferates most readily in nature. As with plants and metazoans, however, this simplistic assumption is tenuous because many of the dominant species in nature do not multiply quickly. In fact, for plants and metazoans, many simple models of species interactions fail because they entail this assumption. The intrinsic multiplication rate of a species in isolation under ideal conditions must be considered along with the limitations imposed by physical and chemical constraints in the environment, the concentration and turnover of limiting nutrients, grazing pressure, competition with other organisms at the same site, and the possible impact of parasites. Given the paucity of knowledge about the impact of these limitations on proliferation, the unmistakable conclusion is that rate of multiplication and, indeed, whether a particular introduction will proliferate at all usually cannot be predicted for most organisms and most environments. Among the few microorganisms that are exceptions are pathogens of humans and economically important animal and plant species as well as microorganisms of environments that are so harsh (e.g., solar salt ponds and hot springs) that few species that reach these sites are able to tolerate the abiotic stresses.

The main attempt to avoid the intensive research effort needed to fill the lacunae described above, and thus to produce a shortcut to predicting survival and initial proliferation, is the hypothesis of "biotic resistance" (Simberloff, 1986b), which states that introduced species are less likely to survive in more diverse, complex communities because of the increased "resistance" of various sorts from resident species. Among possible forces opposing the insertion of an introduced species into a community are competition, predation, and parasitism. Sometimes the survival and effects of a biological stressor seem obviously attributable to a release from biological resistance. For example, the invasion of Lake Huron by the alewife (*Alosa pseudoharengus*) along with its subsequent proliferation and the numerous accompanying ecological effects depended on the prior introduction of the sea lamprey (*Petromyzon marinus*), which greatly reduced populations of native fishes such as the lake trout (*Salvelinus namaycush*) and burbot (*Lota lota*) that would have competed with the alewife (Smith, 1968).

However, it is unlikely that the simple criteria of size and complexity of the resident community will greatly help predict what constitutes a receptive environment. For example, one avatar of the biotic resistance hypothesis is that disturbed habitats are more easily invaded than undisturbed ones. Yet a close examination of records of invasions into various habitat types casts doubt on this view. The habitats, such as agricultural ones, that appear to be particularly invasible are generally anthropogenous ones (i.e., new groups of species in a highly human-modified physical setting), while disturbance per se does not seem to conduce automatically to invasibility. For example, naturally disturbed habitats such as fire disclimax forests or high-energy beaches do not appear to have more introduced species than do other pristine habitats (Simberloff, 1986b). Similarly, another version of the biotic resistance hypothesis is that prior invasion of a system by one or more species hinders successful subsequent invasion. In the systems in which this effect has been studied, however, it appears that the species introduced earlier were a priori more likely to survive independently of which other and how many other species were present, while the later invaders were poorer colonists that would in any circumstances have had a low probability of survival (Keller, 1984; Washburn, 1984; Simberloff and Boecklen, 1991). For microorganisms, apart from some highly stressed or nutrient-poor environments, there is little evidence that species-poor communities are more easily invaded than species-rich ones.

A complication in assessing risk from a novel biological entity is that, even if a site has habitat adequate for survival and reproduction of a species, colonization has a stochastic element such that "replicate" introductions do not have the same trajectory. For example, a few pairs of the Old World house sparrow (*Passer domesticus*) were released in Brooklyn, New York, in 1851, never to be seen again. A larger number of propagules were released there in 1852 with the same result. In 1853, at the same time of year and in the same place, a similar number of propagules were released. The population multiplied enormously, the species spread throughout North America, and it is now one of the most common birds on the continent (Long, 1981), displacing native martins, swallows, and wrens (Sharples, 1982). Similar tales abound among insect introductions for biological control. It is often assumed that such differences in the outcomes of "replicate" releases are caused by genetic differences among the propagules, but such differences are probably unmeasurable for most propagules of potential biological stressors (Simberloff, 1985). Thus, at least some fraction of the apparent stochasticity of biological introductions will always be present.

3.3. Dispersal

Once a species has survived at its site of introduction, its potential spread must be assessed. Many biological stressors are mobile, and their dispersal often does not follow the trajectories associated with movement of chemical or physical stressors. The probability of dispersal or spread of the biological stressor from the point of its first introduction or detection to other sites is critically important to risk assessment. The stressor's impact at the original location may be negligible or even undetectable. If it is transported to a more hospitable site, however, it may cause major harm there. Even if its impact in the environment of initial introduction is substantial, the stress could be highly localized and self-contained unless the stressor spread to new areas suitable for establishment. Moreover, the issue of dispersal is not merely one of physical dissemination. Because the propagule must reach the new locale alive, dispersal must be assessed together with an evaluation of factors that could cause the death of the propagule during dispersal.

Organisms can disperse by a bewildering array of means (Mackenzie et al., 1985; Upper and Hirano, 1991). Particular microorganisms can use one or more of several modes: (1) through the air; (2) in association with currents or by mixing in streams, rivers, lakes, or marine waters; (3) over the soil surface with runoff after precipitation; (4) through the soil with vertical movement of water; (5) through ground water in aquifers; (6) by splashes or raindrops falling on foliage or the soil surface; (7) in connection with animal movement; and (8) as a consequence of human activity or implements. In addition, microorganisms have the capacity to move by such means as ballistic discharges as well as via hyphal growth by fungi, phototaxis by algae in surface waters, or motility by bacteria; the distances reached by such means, however, are rarely more than a few centimeters—a few meters at most—and thus such movement is not likely to be consequential in risk assessment.

Aerial dispersal is often the chief or only dispersal mode of a microorganism (Upper and Hirano, 1991). Individuals may be lifted into the air when winds dislodge propagules from plant foliage; by ballistic discharges that allow fungal spores to move away from the plant or soil surface with small particles of soil carried aloft by the wind; as aerosols from surface irrigation waters as well as from sewage treatment plants; or as dust from farming, construction, or other human activities. The rapid spread of some fungal diseases of plants, for instance, provides evidence of this means of spread. Aerial dispersal depends on the nature of aerial dissemination of particles, the factors that result in the introduction of microbial propagules into the air, and the death rate of propagules during such transport.

Many microorganisms require water to move (Alexander, 1971). Such movement may be tied to lateral currents in both freshwater and marine environments, vertical mixing in many bodies of water, or the lateral transport of water and suspended soil particles following rainfall or snowmelt. The extent of movement depends on the physical transport of the water, the factors that place the organisms in the path of flow, and survival of the organisms as they are being moved. Although bacteria and viruses also pass through soil with percolating waters, few of the propagules move far because they are sorbed to soil surfaces or retained by physical filtration associated with small pores in soil. Nonetheless, many microorganisms can enter the underlying aquifer by passing through channels. While lateral dissemination of bacteria and viruses may occur with the moving stream of ground water, the distances traversed are short.

Biological vectors often serve as an efficient means to move microorganisms. The vectors may be birds, large terrestrial animals, insects, rodents, fish, earthworms, growing roots, and, probably, zooplankton. The number of microbial propagules borne by such vectors may be small; however, the dissemination often is remarkably efficient because the vector may carry the propagule unerringly to a new habitat that the microorganism can exploit, as is common with insect, bird, and rodent transmission.

Metazoans and plants use most if not all of the means that microorganisms do to disperse. They are transported passively by wind and water, carried phoretically by animals, and are often moved by human activities such as agriculture. Movement through soil is relatively unimportant and, in any event, would be quite local. Even passive movement by animals and plants is often initiated, just as in microorganisms, by a behavior or process that places the propagule in the vicinity of the transport agent (e.g., plants explosively release their seeds to wind and water currents, spiderlings assume postures and spin silk threads that facilitate ballooning). Virtually all animal and plant species have characteristic stages in their life cycles during which dispersal is especially likely, and these stages are often associated with behavior or structures that enhance dispersal (Johnson, 1969).

Many animals, however, have more active, sustained behaviors that enhance the probability of great dispersal. Some are migratory, for example, and others have innate behavior that causes their departure from the natal area to be prolonged and, often, to terminate far from it. Further, many animals and some plants can time their movements to increase the likelihood of dispersing effectively and terminating the dispersal in a habitat suitable for existence and reproduction.

Many of the factors governing the transport of inanimate materials, especially particles, would likewise govern the transport of microorganisms and small plants and metazoans. Indeed, much of

the modeling of such transport is based on particle transport or related models, such as smoke-stack diffusion. A critical difference, however, is that living organisms can die (or viruses can lose infectivity). Particularly for passively transported organisms, dispersal models that do not account for the decline of viability or infectivity will overestimate the distance likely to be dispersed or the number of viable propagules that will arrive at a new site. Unfortunately, there are not substantial data on death during dispersal in various media.

Distances traversed by microorganisms, plants, and metazoans range from a few centimeters to thousands of kilometers. In particular, active or passive aerial dispersal (including transport by birds) can move organisms vast distances. The literatures of agricultural and forest entomology, public health, veterinary medicine, and plant pathology provide considerable data on distances and means of spread.

Faced with the bewildering array of dispersal means and their varying efficiencies, Pielou (1979) distinguished between two rather distinct forms of spread. In the first, which she called diffusion (cf. Hastings, 1994), a species' spread more or less closely approximates increasing concentric circles for which the circumferences become progressively more warped. The Colorado potato beetle (*Leptinotarsa decemlineata*) in Europe (Nowak, 1975) and the Japanese beetle (*Popillia japonica*) in the United States are good examples. The rate at which such species' ranges expand is a function of the dispersal and behavioral biology of the species. Various models, beginning with simple diffusion models (e.g., Upper and Hirano, 1991; Strauss and Levin, 1991; Anderson and May, 1991), describe the process. These models typically consider the initial density of organisms and the rate and efficacy of the means of dissemination. The larger the source and the hardier the propagule, the greater the probability of a successful dispersal. The most extensively developed models are epidemiological ones concerned with microbial disease agents. Such models typically predict where microorganisms will be transported and how many will arrive alive. Those that relate to pathogens affecting plants are particularly useful for studying the aerial dispersal of spores that lead to disease in economic crops, while those concerned with infectious agents affecting humans are primarily concerned with host-to-host transmission and with the movements of individual affected hosts.

While some of the more recent diffusion models (e.g., Hengeveld, 1989) seem to simulate some observed gradual spreads of introduced species strikingly closely, it is too early to tell if their predictions of current spreads will be accurate. The warping of the range circumference as diffusion proceeds is probably caused by heterogeneities in the physical environment (Simberloff, 1986b), and it is quite possible that, if one knew enough about the habitat requirements of a particular species, a

diffusion-type model could be modified to reflect these heterogeneities. For example, the Atlantic Ocean and Chesapeake Bay seem to have prevented the Japanese beetle from spreading evenly in all directions from its point of introduction in New Jersey. The unsuitability of ocean as a habitat for a terrestrial beetle is easy to deduce, but the influence of other habitat gradients on diffusion dispersal will be subtler.

Cellular automata are a more recent kind of model that, like diffusion models, depicts the local, gradual spread of introduced species (Hastings, 1994). These computer-intensive models represent each spatial location as being in one of a number of states (e.g., high or low population density). At each time increment, the state of each location changes according to the states of nearby locations and rules based on the biology of the species. A stochastic version of a cellular automaton is called a "Richardson model" and allows for chance events to be introduced into each run. Cellular automata are sufficiently recent that one cannot yet say whether their predictions are more accurate than those of the most modern diffusion models.

Models for aerial dispersal of microorganisms are probably useful in risk assessments of nonpathogenic microorganisms. The paucity of information on death, however, will likely affect their utility because different species have vastly different rates of death due to the irradiation, desiccation, and temperature stresses encountered during aerial transmission. Models for waterborne dispersal have received less attention for microorganisms, plants, and metazoans. Among aquatic introduced plants and animals, the spread of many of the most problematic species (e.g., white amur [*Ctenopharyngodon idella*], European carp [*Cyprinus carpio*], zebra mussel [*Dreissena polymorpha*], purple loosestrife [*Lythrum salicaria*], water hyacinth [*Eichhornia crassipes*], and Eurasian water milfoil [*Myriophyllum spicatum*]) has not been modeled in more than cursory fashion despite extensive empirical information. It may seem reasonable to use models that were developed to simulate the transport of chemicals in water to study microbial or small plant or metazoan dissemination; however, the attachment of microorganisms to particles, the likely different death rates of different species of organisms, and differences in death rates between sorbed and free cells suggest great caution should be used in applying transport models of nonliving materials to organisms. Thus, predicting even the gradual, diffusive spread of living organisms is difficult despite a number of possible models.

By contrast to more or less regular diffusion, some introduced species have spread irregularly from the outset or after a short period of circular range expansion. Often several foci arise simultaneously by long-distance "jumps" (Pielou, 1979), each subsequently serving as a base for slower circular growth or yet another long-distance leap. The aphid *Hydaphis tatarica* was restricted

to a small area of southern Russia, apparently by the limited range of its host, Tatarian honeysuckle (*Lonicera tatarica*). It was in the process of spreading gradually westward and had just reached the Moscow region when it was scientifically described in 1935. As the host honeysuckle was planted as an ornamental throughout much of central and southern Europe, however, the aphid's range increased greatly and irregularly, often to areas not contiguous with the original range.

Modeling jump-dispersal to allow prediction seems a far more formidable task than modeling diffusion dispersal. Probably many more propagules actually undergo jump-dispersal than are recorded. Most of them, however, never establish ongoing populations because they either fall in unsuitable habitat or fail to increase for the various reasons outlined above (e.g., insufficient number of propagules). Yet it is clear that many suitable sites are not reached by adequate propagules of the myriad introduced species that have survived and often increased dramatically in many areas of the globe. Because physical forces such as upper air currents can sometimes be identified as likely agents of jump-dispersal, at least direction may be predictable. For other modes, such as the archetypically improbable event of the transport of a seed on a bird's foot (Simpson, 1952), generating useful probabilities for prediction would be much more difficult. Nevertheless, long-distance transport by birds was certainly important in establishing the ranges of many invertebrates and freshwater algae as well as flowering plants (Carlquist, 1974; Pielou, 1979) and so must be a fairly common event.

Human movement of living organisms (anthropochory), both deliberate and inadvertent, usually constitutes jump-dispersal. Recreational vehicles have transported gypsy moth (*Lymantria dispar*) egg masses on land and zebra mussel (*Dreissena polymorpha*) juveniles in freshwater. Innumerable introductions of terrestrial, freshwater, and marine species have occurred in ship's ballast. The ornamental plant industry spread many important species, such as purple loosestrife (*Lythrum salicaria*), while individuals seeking attractive flowers dispersed others (e.g., water hyacinth [*Eichhornia crassipes*]), and forestry activities yet others (e.g., *Melaleuca quinquenervia*). Fish and game departments as well as individual fisherman have completely changed the ichthyofaunas of many areas, including the American West, where in several drainages most of the native species are threatened (Moyle, 1986). Many biological stressors have been introduced through deliberate or inadvertent release of pets. For certain taxa, introduction societies have changed entire faunas. In the Hawaiian islands, at least 70 species of passeriform and columbiform birds have been introduced; many survived and these completely dominate lowland areas, while native birds are now almost restricted to upland native forests (Simberloff and Boecklen, 1991). Where an introduction is deliberate, an initial jump-dispersal can be predicted. For many inadvertent cases of anthropochory,

detailed prediction will be impossible, although heavy use of certain transportation routes will likely generate correspondingly large numbers of jump-dispersals along those routes.

3.3.1. *Dispersal en Masse*

In the United States alone, 2.5 million gallons of ballast water, originating all over the globe, are released every hour (Carlton et al., 1994). Although this practice was originally thought to be innocuous regarding the transport of biological stressors relative to those that could be carried by the soil previously used as ballast, it has become apparent that ballast water is completely changing the marine biogeography of the globe (Carlton and Geller, 1993) and that many major effects are likely. For example, several scores of species introduced in this fashion can be found together fouling floating objects in San Francisco Bay and Los Angeles Harbor. Detecting even major ecological effects is often difficult, however, and predicting which organism will cause what effect is extremely so (as explained in sections 4 and 6). Nonetheless, given that over 3,000 marine species are in transit in the ballast of ships on the oceans on any given day (Carlton and Geller, 1993) and that at least 367 species of plants and animals have been found in ballast water arriving in Oregon from Japan, one can be quite certain that ballast water is a major pathway for biological stressors and that major effects are already occurring.

Based on information about routes taken by oceangoing vessels, one might predict where ballast-borne species are most likely to come from and go to, although this sort of analysis has been conducted only for the Pacific Ocean. Once an organism arrives in a new biogeographic region by such means, one might use the sorts of models discussed in section 3.3 to predict its further spread; however, such initial long-distance movement of an organism clearly would involve an enormous component of chance.

Nonetheless, an assessment of the risk posed by a pathway for dispersal en masse is a worthwhile exercise. The USDA *Generic Non-indigenous Pest Risk Assessment Process* (Orr et al., 1993) encompasses just such phenomena. For example, the risks of introducing forest pests posed by importing unprocessed logs into the United States were estimated, with the subsequent decision to permit the importation of Monterey pine and two other tree species from Chile (USDA, 1993) but to bar larch from Siberia and the Soviet Far East (USDA, 1991). For each assessment, first the species that might be transported in this way were identified and an estimate of the likelihood of such movement was developed for each (i.e., the dispersal part of the assessment). Then, for each species judged potentially harmful, the assessment team estimated the likelihood of various effects (see section

4) as well as the potential costs if they were to occur. In addition to alerting the public and appropriate authorities to the potential problems, the exercise both formalized what is known about the risks and indicated where major uncertainties lie.

4. EFFECTS

A novel biological entity, such as an introduced species or a genetically engineered organism, can affect an ecosystem in numerous ways. Often, deliberate introductions of game animals are said to be desirable on the grounds that they are "filling an empty niche" (Ebenhard, 1988), and inadvertently introduced species that establish especially large populations also are occasionally said to have "occupied a vacant niche." It is notoriously difficult, however, to show that a niche truly is empty (Herbold and Moyle, 1986), and in one sense it seems impossible that there is such an entity. Because ultimately resources are metabolized in some way, if only by bacteria, it is difficult to imagine that a surviving introduced species would not at least affect the topology of energy-flow webs and nutrient cycles (Simberloff, 1991). In any event, even a species that fills a classically empty niche can have an enormous impact on an ecosystem. In 1788, the first English settlers to Australia brought 7 cattle, 7 horses, and 44 sheep. By 1974, cattle alone numbered 30 million and were producing 300 million cowpats daily, which are not removed by native dung beetles and modify existing plant communities, provide breeding habitat for numerous insect species, and in some areas dry and blanket the ground. Three beetle species introduced in 1967 filled this "empty niche" by quickly establishing over large areas and rapidly removing vast numbers of cowpats, working the dung into the soil. Their full impact has yet to be calculated but must be enormous.

Perhaps, rather than using terminology related to niches, it would be useful simply to consider whether newly introduced species or genotypes with characteristically and radically different ways of "making a living" than any of the resident species are particularly likely to have major impacts. For example, the destruction of ground-nesting endemic birds by predatory mammals introduced to many oceanic islands could easily have been predicted given that these islands had supported no mammals other than bats and no other species "substituting" for predatory mammals in such a way as to affect populations of these birds (Simberloff, 1994a). Perhaps the profound impacts of the zebra mussel (*Dreissena polymorpha*) in the United States (Garton et al., 1993) can be at least partially explained by the fact that this species has two traits that are extremely rare among freshwater mollusks in North America, although common in marine species: planktitrophic larvae and byssal threads for attachment (Johnson, 1994).

This sense that the major effects of some invasions might have been predicted from careful consideration of traits of the invader has led to persistent attempts to predict the effects of biological stressors by compiling lists of traits that predispose the species to invasiveness. Although the results of such attempts, which were in their heyday in the 1960s (e.g., several papers in Baker and

Stebbins, 1965), are still occasionally published, this approach has largely been discarded. One problem is that such lists, or at least readers who sought to use them, often confounded the traits predisposing toward effective dispersal, traits predisposing toward establishment, and traits conducive to major impact on the resident community once established.

However, the main reason that this approach has fallen into disfavor is that one could always find exceptions to such lists—for example, plants with "weedy" traits that are not weeds, plants lacking weedy traits that are very problematic weeds (Mack, 1994). The reason such exceptions abound is doubtless that the kinds of interactions an introduced species can potentially have with the resident community are so numerous. Moreover, many of these rest so heavily on idiosyncrasies of one or more species that they would never have been considered with respect to invasion effects. Thus it is simply not feasible with a simple listing approach to make more than a first stab at predicting invasion results (references in Simberloff, 1989). As has been pointed out by J. Carey (personal communication, 1994), an analogous approach in public health would be to attempt to predict the nature and effects of a pathogen by studying its structure and molecular makeup without looking at infected individuals.

Although largely rejected by ecologists, the attempt to use short lists of species traits to predict ecological effects is by no means a dead issue for workers in related fields. For example, the first five petitions submitted to USDA for unrestricted commercial introductions of genetically engineered crop plants all cite the list of traits conducing to weediness compiled by Baker (1965) as evidence that the engineered plant will not engender untoward effects (Kareiva et al., 1994).

The "biotic resistance" hypothesis, described in section 3.2 with respect to establishment and proliferation of an invading organism, has been equally applied to attempts to predict effects, at least partially in response to the numerous problems with the list approach. However, although laudable in that it focuses on the recipient community, it too has proven to be not especially predictive, doubtless because the single trait of species richness is simply too crude a characterization of a community (Simberloff, 1986b, 1989).

In general, attempts to predict the ecological consequences of biological stressors are confounded by the enormous number of possible effects, whether the effects are on particular species or on the function of entire communities. A biological stressor can have a major impact on a single population (e.g., if it is a host-specific pathogen or parasite), or it may harm several species (e.g., if it has a wide host range). It may upset entire communities, or it may alter one or more processes important to ecosystem function. As noted in section 6.1, one's conception of how important such alterations

and upsets are rests heavily on one's view of the significance of communities and ecosystems as units of biological organization. Although biologists and environmental scientists often have favorite species or processes they feel should be examined in any assessment of ecological effects of biological stressors, many of the favored species or processes do not seem always to have a role so critical as to warrant particular concern. Unfortunately, no consensus has been reached on which species or processes should be singled out for assessment, nor has research been designed to facilitate such decisions. It is almost platitudinous to say that the perceived importance of the ecological consequences of a new species depends on the perceived importance of the species or processes affected. Yet the problem often is lack of agreement on which are the important species or processes.

4.1. Types of Effects

One can begin to enumerate the effects that introduced species may have by dividing them into direct and indirect effects. A species can affect another directly by, for example, killing it, eating it, or displacing it. On the other hand, the effect can be indirect—for example, a species can modify the habitat of another species or reduce its prey. Although such indirect effects often are subtle and difficult to elucidate, they are increasingly recognized as extremely important (Strauss, 1991). They are treated further in chapter 5 on effects characterization.

An introduced species can directly affect another as a pathogen or parasite (Anderson and May, 1982). Myxoma virus introduced from South America into Australia in 1951 initially killed over 99 percent of the huge European rabbit population, though subsequent evolution of both virus and rabbit has allowed some recovery (Krebs, 1985; Williamson, 1992). Frequently, various epidemiological models, often of the diffusion type, can predict more or less accurately the geography and time course of the initial spread of a pathogen (Anderson and May, 1982; Dobson and May, 1986). The evolution of benignity and resistance, however, can severely limit the longer-term predictive power of these models. Further, jump-dispersal of infected hosts can be a more important mode of spread than diffusion.

The impact of a pathogenic stressor can range from insignificant to devastating on the plants or animals it invades. A particular microorganism may multiply on or within a host plant and cause little harm, or it may cause overt damage to a small or large percentage of individuals of a susceptible host species. If that host fills a "keystone" role (Simberloff, 1991) in the community, the microbial stressor could markedly disturb the entire community. Such major disruptions are well documented for several fungi (e.g., those causing chestnut blight or Dutch elm disease). The diseases caused by

these fungi have had major impacts on community structure and ecosystem function, even though they each acted on a single host species. The affected host was a critical component of plant communities and the effects rippled through populations that interacted with the hosts in various ways. For example, the American chestnut (*Castanea dentata*) comprised 25 percent of the individual trees of many forests (Elton, 1958) and more than 40 percent of overstory trees and 53 percent of basal area (Krebs, 1985). Several insect species that are host specific to the American chestnut are either endangered or extinct because of the destruction of their host by chestnut blight (Opler, 1978). The subsequent effects of the loss of these species on their predators and parasites are unstudied. The oak wilt disease (*Ceratocystis fagacearum*) has increased on many native oak species because the population of red oak (*Quercus rubra*), which is particularly susceptible, increased greatly when the chestnut disappeared (Quimby, 1982). It seems inconceivable that the loss of such a dominant tree could not have affected many other aspects of ecosystem structure and function, although no substantial studies have been performed. Whether the consequences of the disappearance of the chestnut qualify as important effects depends on how one assesses significance (see chapter 2 on ecological significance).

Such impacts will be especially devastating if the host population has had no contact with the pathogen or parasite and thus has no resistance. Indeed, the literature of human and veterinary medicine attests to the marked decline in host populations following exposure to novel viruses or bacteria. The impact also is affected by the genetic heterogeneity of the potential host population, its density, and the stage of its life cycle at which exposure to the stressor occurs. The severity of disease is greater if plants in a community are not diverse and if genotypes in a population are few, as is the case with the devastation caused by fungi in the monospecific cropping systems common in agriculture. Furthermore, if the population of susceptible individuals is dense, the effect of a parasitic microorganism is likely to be far greater than if the host population is sparse (Pimentel, 1985; Levin and Harwell, 1986). The stage in the life cycle of host plants or the age of animal hosts also may alter the impact of the pathogen.

Many dramatic examples illustrate damage to natural ecosystems as a result of the introduction of pathogenic microorganisms. In less than 50 years following its introduction from Asian ornamental nursery material, the fungus *Endothia parasitica* became established on 91 million hectares of U.S. forestland and almost totally eliminated the American chestnut (as noted above). Introduction of the fungus *Ophiostoma ulmi* had a devastating impact on forests containing large populations of American elms. *Phytophthora cinnamomi*, a fungus pathogenic to nearly a thousand plant species, has similarly

affected large wooded areas following its introduction into regions of Australia and the United States in which it was not previously found (von Broembsen, 1989). The introduction of the rinderpest virus into parts of Africa resulted in massive death of cattle and wild ungulates (Holmes, 1982) with subsequent ripple effects on other components of the community such as vegetation and predators (Barbault, 1992). Similar effects of myxoma virus and avian disease organisms are discussed elsewhere in this section.

Competition generally is notoriously difficult to document in the field, and declines of native species said to be due to competition from introduced ones often can be ascribed also to other forces. Without experiment, it is impossible to establish the causes with certainty. For example, the decline of the otter (*Lutra lutra*) in Britain and Sweden in the 1950s was for a long time believed to be caused by competition for space with the introduced American mink (*Mustela vison*), which spread rapidly at approximately this time. More recently, however, the decline of the otter has been shown conclusively to be due to organochlorine pesticides; the otter is more detrimental to the mink than the mink is to the otter (Chanin and Jefferies, 1978).

Nevertheless, sometimes the observational evidence convincingly implicates competition from an introduced species in the decline of a native one. For example, the introduction of barn owls (*Tyto alba*) to the Seychelles islands has coincided with the decline of the endemic Seychelles kestrel (*Falco araea*), and this is probably because of nest site competition (Penny, 1974). The limiting resource does not even have to be used in the same way by the native and introduced species for the native to be harmed. For example, because nest boxes on poles for eastern bluebirds (*Sialia sialis*) in Bermuda are used as perches by the introduced great kiskadee (*Pitangus sulpuratus*), the bluebirds cannot nest in them (Samuel, 1975). In addition to such "resource competition," in which a species detrimentally affects another by preempting a limiting resource, an introduced species can engage in "interference competition" by aggressive behavior. On the island of Oshima, in Japan, the introduced grey-bellied squirrel (*Callosciurus caniceps*) chases the native oriental white-eye (*Zosterops palpebrosa*) from flowers of camellia (Ternstroemiaceae). This interference affects both the bird and the plant because the white-eye pollinates it (deVos et al., 1956).

The prevalence of interspecific competition in nature and the nature of the evidence required to demonstrate it have been controversial subjects in ecology for over a decade (e.g., Lewin, 1983). So it is not surprising that it is difficult to estimate quantitatively the fraction of introductions that result in interspecific competition and the importance of that competition. Ebenhard (1988), admitting that the evidence is often sketchy, found reports of potential competition in a selection of the literature for

9 percent of mammal introductions and 18 percent of bird introductions. One might imagine, however, that literature reports would be particularly likely where there is a possibility of competition. Thus these figures may exaggerate the prevalence of competition among introductions as a whole.

An introduced species can prey on a native animal or be an herbivore of a native plant. Bird species have been eliminated all over the world by introduced rats, mustelids, and feral dogs, cats, and pigs (King, 1984; Atkinson, 1989). The most famous case is that of the lighthouse keeper's cat on Stephen Island, New Zealand. The cat arrived in 1894 and eliminated the entire population of the Stephen Island wren (*Xenicus lyalli*) within a year (Greenway, 1967).

Numerous introductions of insect predators and parasitoids for biological control of insect pests and weeds associated with agriculture have lowered the populations of the latter to stable, insignificant levels (Krebs, 1985). Although such an approach is widely touted as "environmentally friendly," particularly in comparison to chemical control, numerous cases have been documented in which introduced biological control agents harmed or even eliminated nontarget native species (Howarth, 1991; Simberloff, 1992). Sometimes the resulting impact is far from the point of introduction. For example, the cactus moth (*Cactoblastis cactorum*) was introduced to the island of Nevis in the West Indies in 1957 to control various species of prickly pear (*Opuntia*). It island-hopped by its own flight throughout the West Indies, reaching the Florida Keys by 1990 and infecting and threatening the entire remaining wild population of the endemic semaphore cactus (*Opuntia spinosissima*), a candidate endangered species (Simberloff, 1992). Because of the mobility of living organisms, it will be difficult to establish a reasonable scope for assessing the risk of potential ecological effects. Clearly, the host range of a parasite or predator is a key component in this assessment.

Given the difficulty of establishing even the status of species of plants and animals that do not stand out in any way (the vast majority), much less the controls on their populations, and the remarkable sets of circumstances that revealed the existence of a few local extinctions caused by biological control introductions, there is every reason to think that predators and parasitoids introduced for biological control have caused numerous local extinctions and possibly global ones as well (Howarth, 1991; Simberloff, 1992). Similarly, it is not possible to estimate accurately how many previous introductions have inimically affected native species by predation. Ebenhard (1988) found the possibility of such an effect for about one-third of 290 species of mammals that have been introduced worldwide, but this can be only the roughest of estimates. Moreover, the effects of mammals are far more likely to be noticed than those of most other taxa.

An introduced plant can affect native plants by allelopathy. For example, the African ice plant (*Mesembryanthemum crystallinum*) has been introduced into California with devastating impact on native vegetation (Vivrette and Muller, 1977; Macdonald et al., 1989). It is an annual that accumulates salt throughout its life. When it dies, rain and fog leach the salt into the soil, where it suppresses growth and germination of native species. An effect similar to allelopathy can be induced by introduced nitrogen-fixing plants. For example, the Atlantic shrub *Myrica faya* was introduced to young, nitrogen-poor volcanic regions of the island of Hawaii where there are no native nitrogen-fixers. The shrub forms near-monocultures, to the detriment of native plants adapted to nitrogen-poor soils and to the benefit of other exotic plants (Vitousek, 1986).

Introduced species can be vectors or reservoirs of disease to which they are more or less resistant but native species are susceptible. The major reason that so many native bird species of Hawaii have gone extinct and that so many of the remainder are threatened is habitat destruction (discussed below). A key contributing factor, however, is avian pox and malaria vectored by birds introduced from Asia in the late 19th and early 20th centuries (van Riper et al., 1986). These diseases may prevent native species from colonizing otherwise suitable native upland forest in which densities of introduced species are high. An introduced species may even serve as a reservoir for a disease that was not introduced with it. On Puerto Rico, for example, the introduced small Indian mongoose (*Herpestes auropunctatus*) carries rabies, but did not introduce the disease (Silverstein and Silverstein, 1974). It is well known that human diseases brought by Europeans to North and South America, Australia and New Zealand, and various small islands around the world were devastating to many native peoples (e.g., Crosby, 1986). There is every reason to think that other animals were equally affected by species introduced by Europeans, and the same may be true of plants as well. Establishing that a native species is limited in range or numbers by a disease is extremely difficult, and determining the origin and reservoir of that disease even more so.

Probably the most important ecological effect of introduced species is modification of the habitat, because such modification can affect entire communities of species and frequently whole ecosystems. In the 18th and 19th centuries, much of the northeastern North American coast consisted of mud flats and salt marshes, not the current rocky beaches. This change was wrought by the European periwinkle snail (*Littorina littorea*) (Bertness, 1984; Dean, 1988), which was introduced to Nova Scotia about 1840 and has slowly expanded its range southward. The periwinkle eats algae on rocks and rhizomes of marsh grasses. When it is experimentally excluded, algae and then mud quickly cover the rocks, after which grasses invade the mud. Thus the physical nature of the entire intertidal

zone of a large region has changed, with the consequent change of the entire community. Introduced feral pigs (*Sus scrofa*) have similarly modified entire ecosystems by rooting and selectively feeding on plant species with starchy bulbs, tubers, and rhizomes (references in Simberloff, 1991). Further, they have greatly modified soil characteristics by thinning the forest litter, mixing organic and mineral layers, and creating bare ground. In turn, these changes increased concentrations of nitrogen and potassium in soil solution and accelerated the leaching of many minerals (Singer et al., 1984). In some areas, the changes have greatly aided the invasion of exotic plants (Loope and Scowcroft, 1985).

A biological stressor can devastate an entire ecological community through habitat modification even if it does not reproduce. For example, grass carp are widely used to control rooted aquatic plants, often with great success from the standpoint of the target species alone (Ashton and Mitchell, 1989). However, fish introductions for this purpose always have unintended side effects (Courtenay and Williams, 1992). Grass carp are often so voracious that they destroy a large fraction of the vegetation, even nontarget species, thereby affecting the entire community of fishes and invertebrates that inhabit this vegetation. These effects are sufficiently detrimental that environmental managers generally do not want grass carp permanently established, leading to the suggested use of sterile triploids (Thorgaard and Allen, 1992). However, no experimental evidence is available on the recovery of communities devastated by grass carp, and the grazing is often so severe that one might expect extremely persistent effects even after disappearance of the carp. It is also noteworthy that supposedly "sterile" triploid fishes may not be completely precluded from reproducing (Thorgaard and Allen, 1992).

An introduced plant can modify the entire plant community by various means. We discuss above such impacts by a nitrogen-fixer in nitrogen-poor soil and by allelopathy. Fire enhancement can have equally great effects. Around 1900, *Melaleuca quinquenervia* was introduced from Australia to south Florida, where it has displaced less fire-resistant species, such as cypress, over thousands of hectares (Ewel, 1986). Several introduced plants act similarly as fire-enhancers in Hawaii (Vitousek, 1986). Because the fire regime can critically determine the composition of a plant community, and the plant community in turn constitutes the habitat for the animal community, such species that effect great changes in the fire regime can have enormous impacts.

Introduced plant species can modify the habitat by constituting forests where none had previously existed. Along rivers in the arid southwestern United States, salt-cedar (*Tamarix spp.*) and Russian olive (*Eleagnus angustifolia*) have had far-reaching effects because they form new forests (Knopf and

Olson, 1984; Vitousek, 1986). Salt-cedars were introduced in the 19th century, and their deep roots allow them to maintain themselves in situations where other plants cannot, such as on the floodplain of the Colorado River. These forests provide habitat for animal species. Also, salt-cedars transpire so heavily around desert springs that they have replaced entire marsh communities with a monoculture forest. The effects of Russian olive are similar, and it is found further north. While its devastating impact on some bird guilds has been studied, its full effect can only be guessed at. Similarly, although mangroves cover intertidal soft substrates in sheltered tropical bays and estuaries in most of the world, they were unknown in Hawaii, where such sites were unforested. In 1902, seedlings of red mangrove (*Rhizophora mangle*) were planted on Molokai. This mangrove species has since spread to other islands by natural dispersal and perhaps deliberate plantings and now forms frequent forests up to 20 meters high. Although the consequences of this new habitat have not been studied, the effect of this change must be enormous. For example, healthy mangrove swamps drop about 4,000 kg of leaves annually per hectare, and the roots form critical habitat for fishes and shrimp (Carey, 1982) and accumulate sediment (Holdridge, 1940).

Finally, introduced species can hybridize with native ones, potentially modifying the native species in some undesirable fashion or even changing it so much that it would not be regarded as the same species. Introgression from cultivated sorghum (*Hordenum vulgarum*) has rendered shattercane (Sorghum bicolor) and Johnson grass (*Sorghum halepense*) more serious pests (Harlan, 1982). Numerous plants endemic to islands suffer hybridization by pollen influx from related introduced species (Ellstrand, 1992). Introduction of the fish *Gambusia affinis* and *G. holbrooki* for mosquito control has led to their hybridization with a restricted endemic (*G. heterochir*) and threatens the existence of the latter as a separate species (Courtenay and Meffe, 1989). In the Seychelles islands, the local subspecies of the Madagascar turtle dove (*Streptopelia picturata rostrata*) has been destroyed by hybridization with the nominate subspecies (*S.p. picturata*) (Penny, 1974). Similarly, in Japan the native subspecies of the Siberian weasel (*Mustela sibirica itatsi*) has been genetically changed by massive hybridization with the introduced Korean subspecies (*M.s. coreana*).

For plants and metazoans, one can predict the circumstances under which such hybridization is most likely. Of course subspecies of the same species can exchange genes. Species within groups with mainly behavioral means of reproductive isolation are probably more likely to be subject to such a process, since the hybrids are at least fertile and prior selection for a behavioral avoidance would not have occurred (Ebenhard, 1988). Wild congeners of feral domestic species are likely to hybridize; for example, few "pure" polecats (*Mustela putorius*) probably remain in Britain because of

widespread release of the domestic ferret, *M. "furo"* (Ebenhard, 1988). It would be difficult to generate quantitative predictions, however, about the risk of such hybridization or its effects on the species concerned. Many species that are closely related do not exchange genes, either because of a postzygotic isolation mechanism, such as chromosomal incompatibility, or because of a subtle prezygotic one. Moreover, some species that are not closely related have formed hybrids in nature, particularly among plants. The particular genes exchanged can have such varied and unpredictable effects as to defy generalization.

"Lateral transfer" of genes also is possible for microorganisms (Stotzky et al., 1991). It may occur even if the original biological stressor fails to survive. Thus the possibility of hazard remains in the form of genetic information in the absence of the original biological stressor. The species now bearing genetic information coding for the deleterious traits may survive, it may be widely dispersed, and it may multiply. Indeed, the recipient microorganism may be more ecologically fit than the originally introduced individual and thus constitute a higher risk. Most of the information on gene transfer among bacteria comes from laboratory studies using highly artificial test conditions or environmental samples treated in fashions that make them unreal simulants of natural conditions. Many of these studies suggest that bacteria are engaging in a wild orgy, exchanging genes with a frequency that would result in the loss of individuality of the particular species involved. Granted some of the enthusiasm of microbial geneticists is excessive; nevertheless, it is likely that gene exchange does occur in nature. The frequency of transfer of such genetic information, however, particularly of genes important in causing ecological harm, remains totally unknown (references in Regal, 1986). It is also worth brief mention that the movement of genes into a population, with subsequent effects as noted above, need not require the introduction of an entire living organism. Rather, components of living organisms such as pollen and plasmids can constitute biological stressors even in the absence of the organisms in which they originated.

Most of the potential effects we have discussed are direct effects. As noted above, myriad types of indirect effects are possible and could be quite difficult to recognize, much less predict. For example, the large blue butterfly (*Maculina arion*) was inadvertently extinguished in Britain by an apparently unregulated biological control introduction (Ratcliffe, 1979). Caterpillars of the blue must develop in underground nests of the ant *Myrmica sabuleti*. The ant, in turn, cannot nest in overgrown areas. Changing land use patterns and reduced livestock grazing left primarily rabbits maintaining the habitat. Then an ad hoc attempt to control rabbits with the myxoma virus reduced their populations and, consequently, reduced ant populations to the point that the butterfly went extinct. Effects may

be even more byzantine. For example, a mite (*Pediculoides ventricosus*) was accidently introduced into Fiji and attacked larvae and pupae (but not eggs and adults) of the coconut leafmining beetle *Promecotheca reichei*. The mite locally destroyed all the larvae and pupae during the dry season. The adult beetles then oviposited and died, converting the beetle population to one with synchronous, nonoverlapping generations. The consequent absence of larvae and pupae during certain periods caused the mite population to crash, as did the populations of two native parasitoids that had previously controlled the beetle. Mite and parasitoids did not live long enough to persist during the intervals between occurrences of the host stages they required for oviposition. So the beetle population exploded (DeBach, 1974). Who could have forecast such an event? Determining that the cause of the beetle explosion was the mite must have been very difficult, given that the mite population was now very low. Had coconuts not been an economically important crop, this case would still be a mystery.

Since species can interact through shared prey or hosts, shared predators, parasites and pathogens, many types of habitat modification, and a variety of tritrophic interactions, the possible sorts of indirect effects are enormous, and there seems no way to do more than to list some obvious possibilities in each case. Certainly a quantitative estimate of the probability of various effects is currently impossible.

We have focused primarily on effects of biological stressors on resident populations, with some examples of communitywide effects. This emphasis reflects the biases of the invasion literature. We have touched on some effects that act through modification of ecosystem processes, such as nutrient cycles and fire regimes. As we note in section 1, an abiding controversy in ecology is whether the functioning of ecosystems can be understood by analysis of its components or whether ecosystems are holistic entities that must be studied at the ecosystem level. Perhaps the attention paid to various levels of ecological organization by students of biological invasions reflects the stance of most of those students toward this controversy. In any event, additional consideration of ecosystem-level consequences of introduced species is summarized by Ramakrishnan and Vitousek (1989).

4.2. Stochasticity, Inherent Unpredictability, and Dose Response

As noted in section 3.2, there appears to be a stochastic element to whether a propagule survives and reproduces at a site. Similarly, there is variation in the effect a new species has on a community once it has successfully dispersed. For example, different invasions of influenza in the same human populations have very different effects (references in Regal, 1986). Some fraction of this variation

may be explained by even minor genetic differences among the surviving individuals in the different propagules. Data on such differences are not usually available.

Further, even aside from truly stochastic variation (e.g., meteorological conditions during the arrival of a group of propagules) and uncontrolled genetic variation, the dynamics of some very simple models of interactions among species are so "chaotic" that the interactions appear to be random and certainly cannot be predicted with much precision (May, 1987). Even with identical initial conditions, several alternative equilibrium states are predicted to be possible outcomes of introduction of the same species to a patchy landscape (Drake et al., 1993). All of these features of introduced species and their dynamics greatly complicate the ability to predict precisely the effects produced by a particular biological stressor, though they by no means prevent the possibility of assigning great risks to certain species and much lower risks to others.

For microorganisms, another source of variation also is quite unpredictable at present. The relationship between the dose of a toxicant and the response of a test population is important for risk assessments of chemicals. A simple dose-response relationship is not evident in available information on biological stressors. It is evident, however, that large populations of some species will produce an effect, whereas small populations cause no demonstrable change. Dose for a biological stressor is number of organisms, population size, or biomass. A simple test system would allow the establishment of a mathematical relationship between population size of the stressor and the response of a susceptible species, permitting, for example, an analog of an LD_{50}. Despite the many parasites, pathogens, and hosts that have been investigated, either such simple dose-response curves do not characterize biological stressors, or, possibly, there are unstudied heterogeneities (e.g., genetic ones) among the propagules. Nevertheless, the outbreak of plant disease to economically significant extents sometimes can be predicted from knowledge of the number of fungal propagules present at a site (Shrum, 1978). Complicating the establishment of quantitative relationships between exposure and response is the fact that many of the potentially harmful propagules can be viable but not active. The density of viable propagules is not necessarily the concentration that poses a threat, because the percentage of these organisms that are active or will be active in the near future depends highly on the particular environment.

The successful establishment of a microorganism in an environment or the creation of an undesirable effect often requires large populations, and a small introduction will not remain extant for long or will have no impact. In effect, a threshold exists for many microorganisms below which the population is not maintained and a deleterious impact is not observed. Although few microorganisms

have been studied in this regard, it appears that the threshold varies greatly with the particular organism. The clearest evidence comes from studies of the number of bacteria needed for infection of humans: For ingested bacteria, 180 cells of *Shigella flexneri,* 10^5 cells of *Salmonella typhi,* but 10^8 cells of *Vibrio cholerae* and *Escherichia coli* are needed (Collins, cited by Levy, 1986). The reasons for these vastly different thresholds are not generally known; in some environments, they may result from predation that eliminates a small population but leaves sufficient survivors of a large population to cause an effect (Ramadan et al., 1990). In other words, whether the stressor has an impact boils down to whether it survives long enough in large enough numbers.

On the other hand, an organism that is frequently introduced into an environment in small numbers may cause an effect, whereas the same population size introduced infrequently will cause no harm. This difference may be the result of changes in the suitability of the environment and the propitiously timed proliferation of the small numbers. In short, there may be stochastic elements in this phenomenon similar to those already discussed with respect to survival.

5. DEFINING ENDPOINTS

Endpoints for some potential effects of biological stressors are straightforward and do not differ in kind from those associated with some chemical or physical stressors. For example, an introduced species that preys on a native one might be expected to reduce the population of that species, and an appropriate assessment endpoint would be this population decline. The measurement endpoint would then be identical to the assessment endpoint—one would monitor the population. If the assessment endpoint were construed in this instance as some change in community structure or function, again the measurement endpoint might well be population size, or it might be some nutrient or energy flow believed to be associated with the prey species. Many of the ecological effects discussed in section 4.1 similarly would not present unique problems in defining endpoints.

The evolution of both the introduced and native species, however, and the possibility of gene exchange between them (discussed in sections 2 and 4.1) complicate greatly the definition of justifiable endpoints. The evolution of resistance, virulence, and benignity can be very quick, particularly if the species involved have short life cycles. At other times, however, such processes can be lengthy with an end effect that is important nonetheless. Even the coevolution in the myxoma-rabbit system in Australia, which seems especially rapid from a biological control standpoint, would be viewed as quite slow in the context of a typical risk assessment. There were about six epizootics in the first quarter century after viral introduction, and the rabbit mortality rate did not fall substantially until the third one (Krebs, 1985).

The acquisition of novel hosts by introduced phytophagous and parasitic insects will bedevil risk analyses. For example, the American tephritid fruit fly *Rhagoletis pomonella* originally was almost wholly a pest of hawthorn, although apples, introduced from Europe, had been within its range and habitat for almost two centuries. In 1865 it was first recorded as attacking apples in the Hudson River Valley, and this "apple race" spread rapidly to southern New England and beyond (Bush, 1975). Currently, it is an economically important pest of apples in much of the eastern and midwestern United States. Many successful biological control programs entail new associations between parasitoid or phytophage and host, and these associations include species that had been thought to be monophagous or oligophagous (Hokkanen and Pimentel, 1984, 1989). Indeed, the very conditions in which most introduced species find themselves upon introduction (e.g., small population size, novel environment) might be expected to lead to rapid evolution, which may include new host range (cf. Roderick, 1992); moreover, a single gene mutation can change host specificity (Williamson, 1992). Such host range expansions occur in nature, as with the Australian gall wasp

(Dennill et al., 1993), and can be produced by genetic engineering, as has been done in certain fungi (Schaeffer et al., 1989).

Similarly, species can evolve an expanded tolerance of physical factors that would greatly increase the probability of important ecological effects, but this might take a long time. The evolution of resistance to insecticides and herbicides is a well-known phenomenon (references in Begon et al., 1990), sometimes occurring quickly and other times much more slowly. Similarly, plants evolve tolerance of novel soil contaminants at varying rates (e.g., Walley et al., 1974). Although the cause of sudden outbreaks in which available resources seem not to have changed is uncertain, genetic change such as a mutation or a recombinant event is often likely (Simberloff and Colwell, 1984). Because both mutations and chromosomal crossover are random events, it is not possible to predict their nature and very difficult to specify even the probability of a detectable change of this sort. Further, these events can happen at any time, including many generations after the introduction.

Sudden dramatic increases in geographic range also are documented, such as that for the collared dove (*Streptopelia decaocto*) in Europe (Simberloff and Colwell, 1984). While none of these has been linked conclusively to genetic change as opposed to habitat change, it is not impossible that such changes have occurred.

Perhaps the key aspect of possible genetic and evolutionary change that bears on choice of endpoints is that much of it is not incremental, but would arise abruptly and might begin to generate effects almost instantaneously. Thus, whereas the addition of a chemical stressor might be expected gradually to lower the size of one or more populations or gradually to affect a population statistic such as fertility, a biological stressor might have little or no effect for a period, until a specific recombinant event or mutation produces a different sort of organism with different effects.

6. UNCERTAINTY OF BIOLOGICAL STRESSORS AS OPPOSED TO CHEMICAL AND PHYSICAL STRESSORS

Biological stressors, then, present a series of unique challenges to risk assessment. These are largely associated with the complexity of communities of interacting organisms and the likelihood of some degree of evolutionary change, which is at least partially aleatory rather than deterministic. Such complexity forces a risk assessor to consider myriad possible effects—many of them probably important to community function—as assessment endpoints. Additionally, the complexity of community organization means there will be some uncertainty in associating measurement endpoints with assessment endpoints, particularly assessment endpoints at the community and ecosystem levels. Even if a set of measurement endpoints can be agreed upon as relevant, sufficient monitoring will likely entail a heavy economic and work burden because there are so many factors to monitor. The evolutionary change complicates endpoint choice in two ways: It makes temporal limits questionable, and it means that change need not be incremental.

Moreover, quantitative risk assessment models are more difficult to devise for biological than for chemical and physical stressors, because living organisms are quirky and communities are complex. Characteristics of living organisms such as evolution, behavior associated with dispersal, and dormancy are all difficult to model. Even static community models are highly idiosyncratic and their predictions not well borne out; accurate dynamic community ecological models simply do not exist. These are the reasons for our emphasis on kinds of effects and the heavy use of examples, an approach that is probably the best that risk assessment can achieve for biological stressors.

For both introduced species and genetically engineered organisms, forecasts have been attempted on the basis of the ecological effects of similar genotypes or species. One might reasonably doubt, however, that such predictions can be useful as anything more than a "shopping list" of potential effects to anticipate. For example, the distinctly different trajectories of closely related introduced species suggest that the experience of one could not have been particularly helpful in predicting the effects of the other. Consider the tree sparrow, which was restricted for over a century to the vicinity of St. Louis, and the house sparrow, which spread over almost all of North America and is now one of our most common birds (see sections 2 and 3.2). These congeners are extremely similar in both morphology and habits in their native Old World ranges. There is currently no convincing explanation for the dramatic difference in their effects. More puzzling still is that, in parts of their native ranges, the tree sparrow is more common, as it is in Australia, where both species are introduced (Long, 1981).

Similarly, four congeneric mongoose species have been introduced for rat control in various parts of the world. One, the small Indian mongoose *Herpestes auropunctatus,* has survived in most regions to which it was introduced and had a devastating impact on native birds and reptiles (Lever, 1985). The others all failed to establish populations (Ebenhard, 1988). Numerous aphelinid wasps of the genus *Aphytis* have been introduced to California to control the citrus pest *Aonidiella aurantii* (the California red scale), with widely differing results (Simberloff, 1986b). Some have quickly disappeared, others became established but were severely limited in range or density; a few established substantial populations over wide areas but had distinctly different effects on the scale. In this rare instance, intensive research was performed to explain some of the differences in these trajectories (Luck et al., 1982), and the results indicate that no superficial effort would have predicted the different outcomes.

Further, the literature on introductions is highly biased. Almost certainly most introductions fail to establish ongoing populations; of those that do establish, almost certainly most have rather small effects on the target community or ecosystem (Simberloff, 1991). Introduced species that have survived and have had major impacts, however, are much more likely to have been noticed. Probably the more substantial the impacts, the greater probability that the species will be studied and the study published.

It is often claimed (e.g., Brill, 1985; Davis, 1987) that the impact of the introduction of a genetically engineered organism on a target community will be easier to assess than the impact of an introduced species, because the genetic change can be characterized quite precisely. One should view this assurance skeptically, however, on two main grounds: First, the differences between similar species or even similar genotypes that distinguish innocuous from tremendously important components of a community are often very subtle and would not easily have been deduced merely from a knowledge of each species or genotype in isolation. Intensive field research may be needed. For example, of two closely related fire ants in the southeastern United States, the native *Solenopsis geminata* is a relatively minor, unobtrusive species, while the introduced *Solenopsis invicta* is one of the major regional insect pests, affecting plant communities, native insects, and quite possibly vertebrates like gopher tortoises (*Gopherus polyphemus*). Yet the biological distinction that has caused the radical difference in these ecological impacts is probably a very subtle behavioral one that determines the size of the mounds and thus production of alates (Simberloff, 1985). The rice brown planthopper (*Nilaparvata lugens*) demonstrates that even a single gene change can greatly affect ecological impact. Until recently a minor pest of rice, it has become a major pest over much of Asia

since the 1970s, apparently because of single gene changes that affect the efficacy of resistance in rice (Sogawa, 1982). Further, the limited research on the precise determinants of species' biogeographic ranges (e.g., Neilson and Wullstein, 1983) suggests that these determinants cannot be gauged other than by intensive field research, and that they are generally very subtle traits that could be modified easily by a single mutation.

A second reason for skepticism about the predictability of ecological effects of genetically engineered organisms is the possibility of lateral transfer of genes from one species to another (see section 4.1). Although it seems possible through extraordinary measures to prevent such an occurrence in a small-scale field test of plants (Gillett, 1987), such events would not be controllable in nature. Too little is known about such genetic exchange in both plants and bacteria to assume that it will not happen occasionally or that its effects will be minimal (Tiedje et al., 1989).

6.1. Action in the Face of Uncertainty

Decisions will be made even in the absence of good science and adequate predictive capacity. Thus, guidance must be provided to the risk assessor, but it should be used only after the enormous uncertainties involved are noted. As pointed out in section 3, predictive ability or knowledge of four major factors is needed to perform a risk assessment: probabilities that the biological stressor will survive, that the survivors will multiply, that propagules will disperse, and that the organism will be harmful. With some introduced organisms, information about one or more of these factors will be extensive; more often, such information will be scanty. It may be possible to draw on analogies with closely related organisms. For example, the U.S. Department of Agriculture and the U.S. Fish and Wildlife Service rely on such analogies in their risk assessments of introduced species and genetically engineered organisms. Such information should be used, however, only with an explicit acknowledgment that related organisms often behave in vastly different fashions with respect to one or more of the four factors.

Often a Delphic process is used to provide essential information on biological stressors. This approach calls on committees of experts on related organisms in an attempt to assess the nature of risks from a new species (e.g., the blue-ribbon committee at the University of California, discussed in section 2, which was convened to assess the risks posed by the Mediterranean fruit fly in California; the National Institutes of Health Recombinant DNA Advisory Committee). The members of such committees generally are "experts" in the sense that they have extensive professional knowledge about related organisms. Even though they are not able to muster evidence beyond what is available in the

scientific literature, their "feel" for a group of organisms makes the Delphic approach the correct one for risk assessment. Because the Delphic approach requires a team, however, which must be composed of experts, risk assessment for biological stressors using this approach is a burdensome process. Nonetheless, no other method is currently available for providing adequate insight into these risks. The risk assessments for importation of Siberian and Chilean logs discussed in section 3.3.1 (USDA, 1991, 1993) give an indication of the size of the team and the depth of expertise required.

Of course, an expert's "feel" cannot substitute for experimentally tested and scientifically derived knowledge about a system. When asked to deal with applied environmental problems, ecologists typically claim that further basic research is needed. While this need is perhaps greater in regard to risks from biological stressors than for most other environmental problems, it is worth noting that a vast literature already exists on several factors that are important for risk assessment, and an expert team would have access to this literature. Unfortunately, much of that information does not deal directly with the organisms of likely concern to risk assessors, except in cases where the stressor is a well-known pathogen of humans, livestock, or crops. Further, almost all potentially relevant cross-genus or cross-family generalizations are riddled with exceptions. Thus, one research need is information on the specific morphological or physiological characteristics that are the basis for survival, multiplication, dispersal, and ecological effects. Currently available information on a number of species will have to serve for now as a generic basis for predictions of risk. Thus, until the amount of such information increases significantly, risk assessment for biological stressors will be speculative at best.

For the risk assessment process addressing biological stressors to be scientifically defensible and transparent, it is crucial that assessors acknowledge the levels of uncertainty. For example, the USDA's generic risk assessment process (Orr et al., 1993) mandates declaring how certain the expert team is about each step in an assessment, and it provides an algorithm for combining the uncertainties to give an indication of the uncertainty of each project as a whole. This approach is illustrated in the assessment for Chilean logs (USDA, 1993).

7. RECOVERY

A major issue in assessing ecosystem recovery (see chapter 7) is the "moving target" problem (Simberloff, 1990a). Communities, and the ecosystems in which they are embedded, are dynamic entities, so they change to some extent even in the absence of any stressors. Indeed, even climax communities are never truly unchanging. Therefore, to determine whether a complete true "restoration" has occurred (i.e., reestablishment of the community that would have obtained had the stressor never been introduced), one would have to know the trajectory that an ecosystem would have taken in the absence of the stressor. Because such knowledge is imperfect, the best one can hope for is that the system's structure and function are not outside the bounds established by normal, dynamic ecosystem and community processes. Another issue concerns the level at which one assesses restoration. If one weighs various ecosystem properties, such as those related to energy flow and nutrient cycling, one can claim restoration much more easily (aside from the moving target problem) than if one demands population-by-population matching of the prestressor ecosystem and the stressed one. Often the goal of recovery may be less than complete restoration, however—that is, simply the establishment of some community with specified characteristics or of an ecosystem with certain functions may suffice ("replacement" in the terminology of Bradshaw [1987]).

Recovery from a biological stressor can differ significantly from recovery from chemical and physical stressors because of several irrevocable changes that can be brought about by a new organism (Simberloff, 1990b). This will be particularly true if one seeks full restoration, on a species-by-species basis, rather than simply recovery of ecosystem function. Of course, complete restoration of structure is impossible if the stressor causes global extinction of a resident species, or it may take a very long time if local extinction must be redressed by immigration from a distant population. The opposite effect is probably even more problematic—it is tremendously difficult to eliminate an introduced species once it has become established (Simberloff, 1994b). Even where a sustained, regional effort has succeeded in eradicating an introduced species, as with the coypu in Great Britain (references in Simberloff, 1990b), restoration has been slow. In some cases, it will be impossible. Virtually all the endemic forest birds of Guam have been eliminated by the Australian brown tree snake (*Boiga irregularis*) introduced in the late 1940s or early 1950s (Savidge, 1987). Even if the snake were removed, which seems a distant dream today, in what sense could one ever say that the island community has "restored"? It will never again have the species of birds that it once contained.

A genetic change in resident species caused by biological stressors, even stressors that do not persist, might further complicate recovery. Moreover, hybridization, lateral transfer, and even "ordinary" evolution, such as resistance to a new disease or parasite, will change resident species permanently.

Nonevolutionary effects of biological stressors, such as habitat modification or various forms of population suppression (e.g., predation, parasitism, competition), resemble effects of chemical and physical stressors. Yet because introductions are usually irreversible—at least given current technologies (Simberloff, 1994b)—assessment of recovery from biological stressors usually differs from assessment of recovery from chemical and physical stressors. Thus this type of recovery assessment will have to account for the predicted continued density of the biological stressor.

8. SUMMARY

Four factors—survival, multiplication, dispersal, and ecological effects—determine the risk posed by a biological stressor. For particular classes of biological stressors, some information exists for one or more of these factors, but little or no relevant information exists for many potential biological stressors. Thus risk assessors often are drawn to examine species that are closely related to the stressor, an exercise that requires recognition of the significantly different ways in which closely related species behave.

The fact that biological stressors reproduce and may multiply renders risk assessment for such stressors more difficult than for chemical or physical stressors. The assessment can be further complicated by the myriad ways that biological stressors disperse, both actively and passively. Particularly noteworthy is the frequency of jump-dispersal in comparison to simple diffusion. Although the exact trajectory of such discontinuous dispersal is extremely difficult to predict, it may be possible to assess the probability that it will occur and to assign greater probabilities to certain geographic routes than to others. One factor to consider is that jump-dispersal is characteristic of but not restricted to transport of living organisms by humans.

Once a biological stressor reaches a site, its potential effects on the target ecosystem are numerous. Since many types of effects have been extensively studied by ecologists, the ecological literature can be useful for beginning to assess potential risks. Studies may indicate, for example, that one should be particularly vigilant about anticipating certain types of effects. Nonetheless, for most species of potential biological stressors, the literature is deficient concerning effects.

Perhaps the key difference between biological stressors and chemical or physical ones—the factor that most affects risk assessment of a biological stressor—is that biological entities evolve. Because evolution can play a role in survival, multiplication, dispersal, and effects, it can have a profound influence on the risk posed by a particular stressor. Because it is a stochastic process, however, and because many aspects of evolution remain quite unpredictable, performing a risk assessment for a biological stressor can be quite complicated. At present, a Delphic process remains the most reasonable starting point, once the uncertainties involved are clearly stated.

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GLOSSARY

Allelopathy—The influence of plants on each other caused by products of their metabolism.

Compound property—A property of a level of organization that is defined only at that level of organization but that follows directly from properties of component entities at a lower level of organization. For example, community respiration is a compound property of a community and is the sum of the respiration of all individuals in the community.

Diapause—A period of suspended growth and reduced metabolism in the life cycle of an organism.

Disclimax—A community that would have been replaced in ecological succession by a different community except for the continued action of a disturbance, such as fire.

Emergent property—A property of a level of organization that is not possessed by lower levels of organization and is not determined trivially by properties of components.

Metapopulation—A group of separate but interacting populations, each with a finite lifetime. The nature of the interactions is not specific.

Phoresy—The transport of an individual organism by another, as seeds in the feathers or fur of an animal.

Planktitrophic—Feeding on plankton.

Population—A group of interacting individuals. The nature of the interactions is not specific but usually includes gene exchange.

Propagule—An individual plant, animal, or microorganism that colonizes a new site. The life-cycle stage of the propagule is not specific.

Issue Paper
on

ECOLOGICAL RECOVERY

Stuart G. Fisher
Department of Zoology
Arizona State University
Tempe, Arizona

Robert Woodmansee
Department of Range Science
Colorado State University
Fort Collins, Colorado

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U.S. Environmental Protection Agency

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1. INTRODUCTION

1.1. A Complex of Considerations

Ecosystems, whether "natural," extensively managed (e.g., replacement of a few of the native species) or intensively managed (e.g., massive replacement of species and disruption of soils and sediments), have been described in terms of many biological and physical variables (e.g., biodiversity, energetics, patterns of nutrient cycling). Recent descriptions include aspects such as physical, chemical, and biological services; economic and social values and perceptions (i.e., productivity, profitability, sustainability, and equitability); and potentiality (i.e., values of future and former states). While all of these characteristics represent legitimate viewpoints and must be considered in ecosystem management, protection, restoration, and rehabilitation, the ability of ecosystems to respond to stress and disturbance, whether natural or human-caused, is often overlooked. If anticipated and desirable, the response process of changes in biological and physical expression is termed recovery; if the response is deemed undesirable, it is termed degradation or decline. As a property of ecosystems, recovery or the potential for recovery should be evaluated during risk assessment.

While the *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) does not explicitly incorporate an analysis of ecosystem recovery, failure to consider recovery can lead to faulty assessments by either overestimating or underestimating the risk involved in any ecological situation. For example, although restoration of a forest to climax (or stability) stage may appear to be complete, damage to seed banks might impair the capacity of the ecosystem to recover from future disturbance events. Loss of refugia or interruption of linkages that may not be evident in normal ecosystem functioning may be central to homeostatic response to stress. Thus, the capacity of an ecosystem to respond to disturbance must be considered an important assessment endpoint.

Ecosystems are always in a state of change, responding to environmental stresses and disturbances at short and long time scales. These stresses may be predictable (e.g., diel and seasonal), unpredictable (e.g., volcanic eruption, landslides), or somewhere in between (e.g., flooding, drought, fire). They may be natural or human-caused, or they may reflect the interactions of factors. Stresses can cause the existing ecosystem to change permanently to another state (i.e., into a different ecosystem) that may not be prescribed or desirable (degradation). Or they can cause an ecosystem to be altered temporarily, before returning to its previous state (recovery). Yet another outcome of change can be the development of a prescribed and desirable ecosystem state that is distinct from the previous state.

Recovery, or succession (i.e., orderly ecological change leading to an anticipated and desirable endpoint), is an ecosystem property that can be utilized to hasten the return of an ecosystem to a selected endpoint, providing an ecological service that mitigates damage. By understanding the factors involved in successional change and using them in the assessment, the manager may facilitate return to the desired endpoint (e.g., by reseeding with pioneer species or delaying fish stocking plans). The ecological risk assessment should factor in the ecosystem's natural capacity for recovery.

The potential for an ecosystem to recover depends on its current state, its current stressor and disturbance regimes, its successional history, its history of disturbance, the desirable future state, the intensity of management applied, and chance (figure 7-1). Establishing desired outcomes of recovery (i.e., endpoints) requires consideration of the current state, some desired historical state, or, in some cases, a concept for a more desirable state than currently exists or previously existed. All of these factors require careful evaluation given uncertainties about ecosystem properties (see chapter 8, on uncertainty in ecological risk assessment). For example, susceptibility to a stressor may be a function of successional stage, in terms of the species that is dominant when stress occurs, the total biomass of the system, or community seasonal phenology. Moreover, the state of recovery can influence patterns of exposure and differential contact with components of the ecosystem. For example, toxicants released shortly after flooding in a lowland stream may be routed quickly through the system; some months later, however, when macrophytes dominate the benthos, the residence time of toxicants may be greatly increased. Thus, the successional (recovery) status of the target ecosystem must be considered when assessing the ecological effect of stressors.

Disturbance events that occur in sequence can have varying effects because recovery from an earlier disturbance may not be complete before the onset of subsequent events. Thus, risk to the ecosystem can be quite different if natural disturbances have been frequent and severe as opposed to infrequent and benign. Moreover, simultaneous agents (e.g., anthropogenic and natural) can exacerbate or mitigate the effect of individual stressors, depending on the successional status. Because sequential stressors can be natural or anthropogenic, their analysis must take all agents into consideration. Thus, stressor effects must be considered in regard to regimes of disturbance and the related recovery state of the ecosystem.

Because typically ecosystems are in the process of recovering from natural disturbance events, a range of states is available for endpoint selection by the risk assessment team. Because a risk assessment requiring restoration of a climax state for an ecosystem that was far from this state when the stress occurred would be unrealistic, a better restoration goal might be an achievable state on the

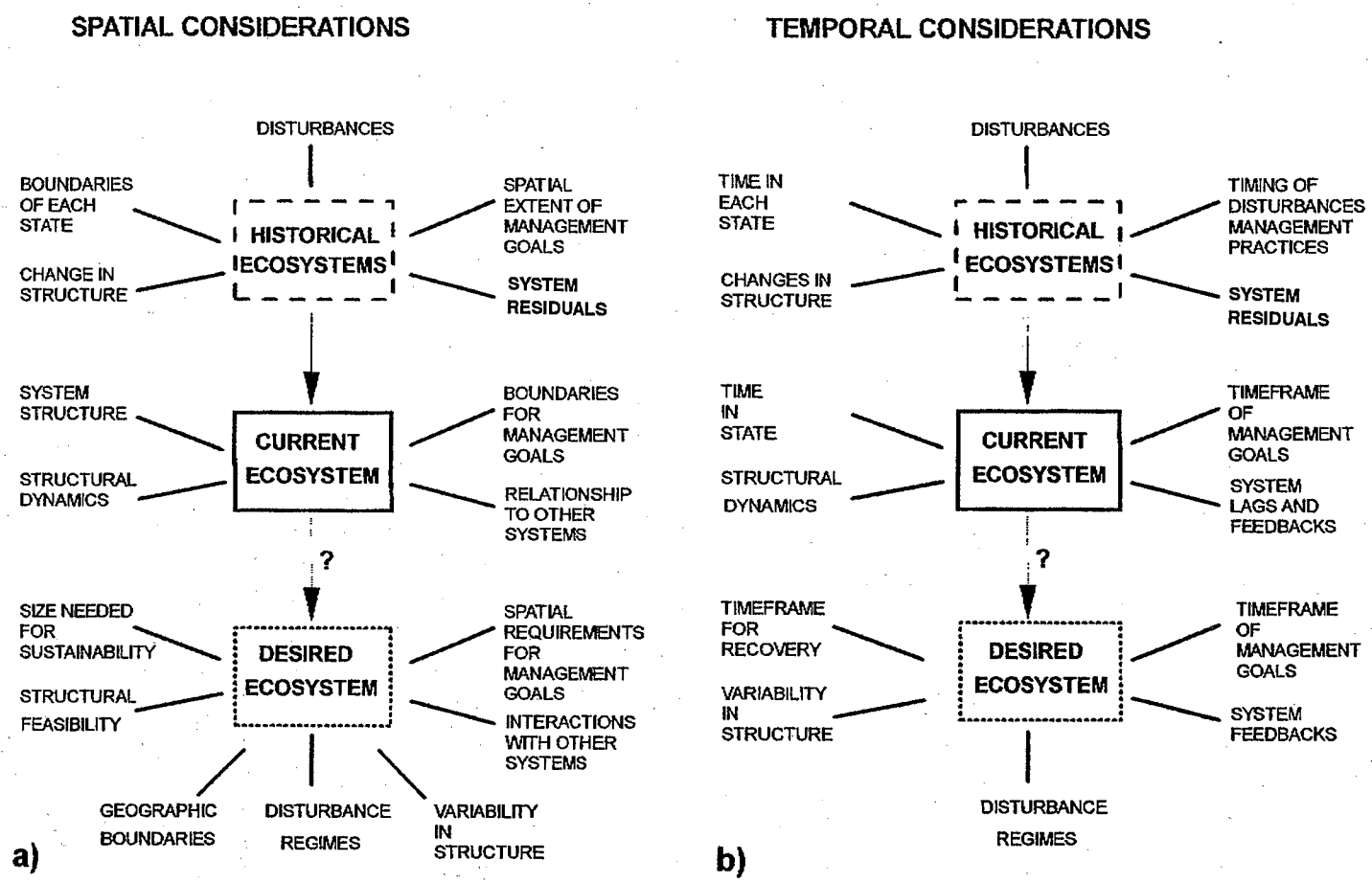

Figure 7-1. Spatial and temporal considerations in describing ecosystems.

recovery trajectory. Thus, an appropriate alternate goal might be a somewhat earlier or later successional state than the one exhibited by the ecosystem when the stress event occurred. To chose the best endpoint (assuming sufficient knowledge exists to make the choice), it is necessary to know the course of the ecosystem's natural trajectory (i.e., to understand successional pathways exhibited by the particular ecosystem under study). Ultimately, the goal should be to get the ecosystem back on track.

Choosing the most desirable ecosystem outcome also involves establishing a social contract based on management goals, policies, laws, and regulations. While directing ecosystems along desirable pathways is the responsibility of, for instance, land and water managers, assessing which of these pathways is the most probable is the considerable task left to the risk assessment team. This chapter focuses on the ecological implications of ecosystem recovery following disturbance that the risk assessment team must consider. Social, institutional, political, and economic factors are discussed briefly because most, if not all, ecosystems are directly or indirectly managed or significantly influenced by humans. In our view, these human factors cannot be ignored.

1.2. A Protocol for Recovery Analysis

This chapter is organized in part around a suggested protocol for assessing the potential for ecosystem recovery and raises some essential ecological and social considerations. This road map (figure 7-2) is adapted from the Structured Analysis Methodology being developed by the Terrestrial Ecosystem Regional Research and Analysis (TERRA) Laboratory (DeCoursey et al., 1993; Woodmansee and Riebsame, 1993), which represents emerging structured analysis schemes intended to facilitate knowledge management and interaction within and between groups.

Developing a clear problem statement that describes the potential for recovery of an ecosystem is essential for performing a realistic assessment (see section 2). Because the effects of recovery might be felt by many sectors of society, the problem definition phase should not be exclusive to risk assessment teams and scientists. We emphasize the need for agreement among people ("stakeholders") with differing viewpoints (figure 7-3) about the nature of the anticipated impacts and the potential recovery; the policy and management implications associated with the recovery; issues concerning who will define policy and management goals, make and implement the choices, bear the costs, and realize benefits; and the biological, physical, economic, and social constraints that are operable. Along with developing the fundamental problem statement, the spatial and temporal scales of the problem must be established. Next, the basic assessment, or conceptual modeling phase of the

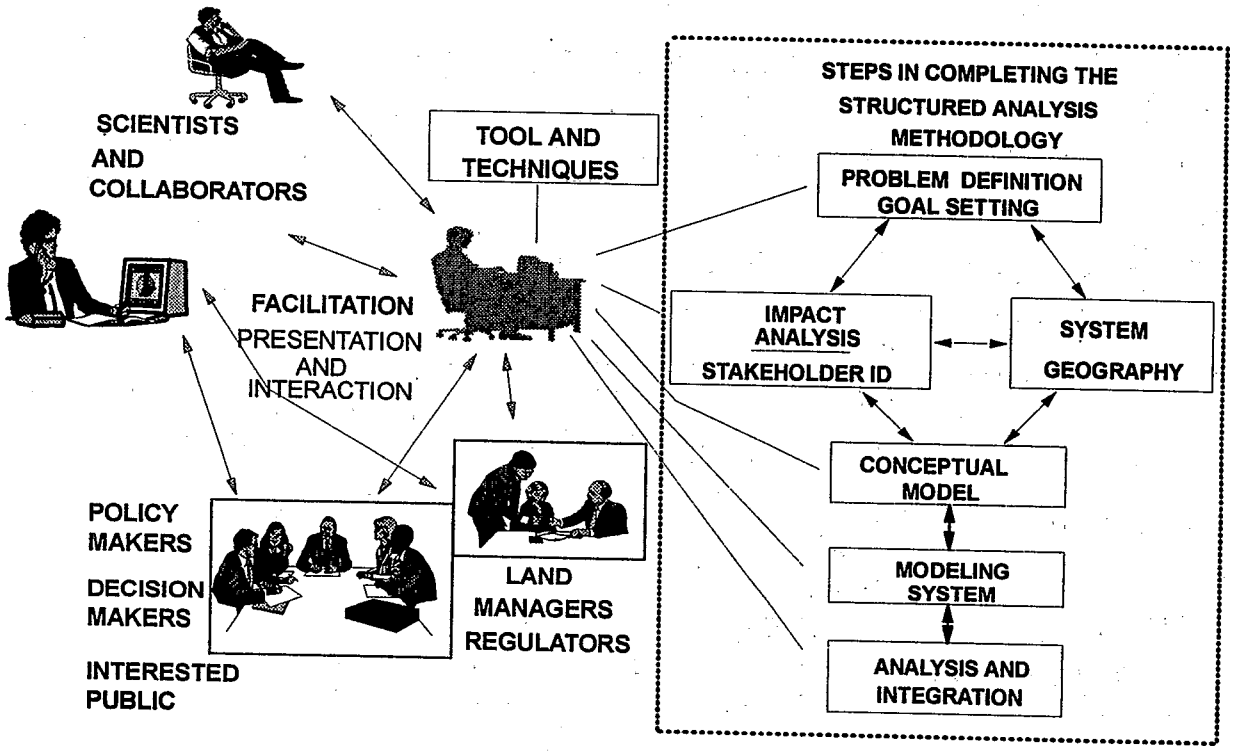

Figure 7-2. Recovery analysis protocol (adapted from DeCoursey et al., 1993; Woodmansee and Riebsame, 1993).

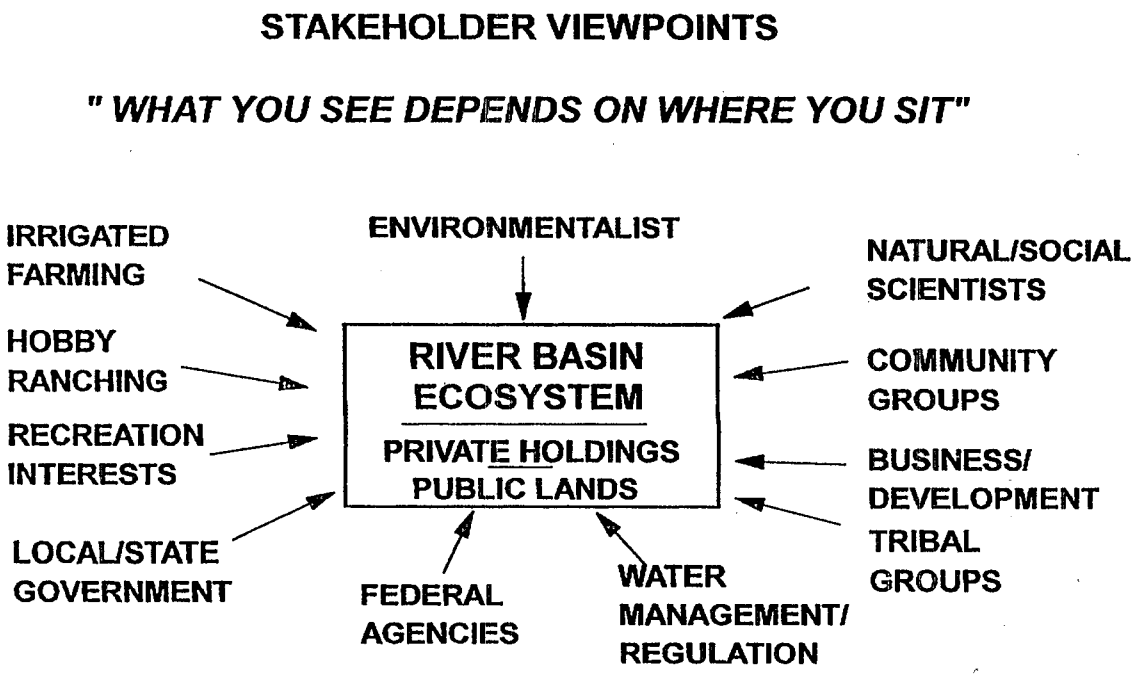

Figure 7-3. Generic river basin.

analysis, can be accomplished (see sections 3 and 4; related considerations are taken up in section 5). The assessment must be based on strong theoretical foundations in the natural and social sciences as well as the most complete information available on the specific ecosystem relevant to the problem statement. Further, available information from the most reliable ecological, economic, and social/cultural databases and the most relevant expertise must be included.

The final steps in the protocol are beyond the scope of this paper. Upon completion of the conceptual modeling phase, a formal mathematical or expert systems modeling effort should be established, if resources permit. Next, empirical studies, intuition, and common sense can be used in analyzing the integration of the preceding steps. Finally, scientists, managers, policymakers, and representatives of the public need to debate and discuss issues and then achieve a consensus on recommended management strategies for achieving recovery. If properly structured, such a protocol can establish clear linkages to risk management activities.

We recognize that implementation of this protocol is dependent on available human and financial resources, the availability of information and, indeed, basic knowledge, as well as the urgency of the problem, the political interest, and the institutional mandates. Yet, if ecological risk assessment is to include considerations of ecosystem recovery, then such a structured analysis is necessary, even if the formal modeling and intensive data analysis phases need to be neglected.

2. PROBLEM DEFINITION AND ESTABLISHMENT OF THE SPATIAL AND TEMPORAL SETTING

Analysis of the precise effects of any disturbance on an ecosystem calls for rigorous description of its current state, its history, the nature of the proposed or existing stress, and the nature of the subsequent management system. Information with which to develop such descriptions may not always be available, however. At the least, to establish the clearest possible definition of the problem, the risk assessment team, risk managers, and principal stakeholders must agree on the problem's fundamental nature within the context of existing knowledge. In particular, the geographic and time-related boundaries and dimensions of the problem must be specified (i.e., Will recovery be expressed at the regional, landscape, or patch scales over periods of seasons, years, decades, or centuries?). Further, as the spatial dimensions of change are specified, the appropriate conceptual hierarchies (i.e., biological, physical, economic, and social) must be established (Woodmansee, 1989; Rosswall et al., 1988) along with description and quantification of important factors within each area where possible. Assumptions about important but poorly understood factors must be clearly stated.

After describing its current state, the analysis must consider the ecosystem in the terms of its successional status (e.g., Does its state represent a major departure from "natural" seral stages? Is it a completely exotic system? Has natural disturbance history or management significantly altered its successional trajectory?). Clear descriptions of these historical considerations are essential for evaluating the recovery potential of any ecosystem.

Finally, the probable effects of the disturbance or stress from which recovery is desirable must be described (e.g., Is the disturbance or stress equivalent to naturally occurring analogs? Is it an exotic stress?). Since such issues are discussed in other chapters, this chapter emphasizes the importance of the stressor's nature in assessing the potential for the recovery of an ecosystem.

2.1. Who Defines the Problem and the Recovery Endpoints?

Since contemporary ecosystem management and regulation schemes emphasize hierarchical decision-making processes, many systems are managed by a centralized authority, such as an owner/operator, a governing board, or a legally appointed administrator. Similar to this type of management approach is decision making by institutional mandate. For all such organizational entities, the lines of authority are direct and decisions are absolute. Explicit and implicit regulations and policies, which are derived and interpreted in myriad ways, influence management decision making by organizations. Decisions usually are consistent with the mission and mandate of the

organization responsible for regulation or management of the specific ecosystem. Such contemporary organizational mandates tend to be narrow, singling out commodities or species, for example, or they emphasize service. The same can be said about environmental impact assessments—and will be said about ecological risk assessment—if old patterns are maintained.

Increasingly, these conventional approaches to decision making are being challenged by various sectors of society. Trends are indicating clearly that authoritarian management is no longer acceptable, as individuals and "stakeholder" organizations seek greater participation in the regulation and management of the environment (Naisbett, 1982; Naisbett and Aburdene, 1990; Peters and Waterman, 1982; and Kessler et al., 1992). As a result, society has placed a great deal of decision making and regulatory responsibility in the hands of judges. The courts, however, are adversarial by design and oriented to unambiguous resolutions, while few environmental issues are that simple.

A far more sensible approach to establishing regulations concerning recovering ecosystems is to bring interested individuals and organizations into the risk assessment process and, where possible, make them part of risk assessment teams. In an inclusive approach, conflicts can be resolved and a consensus can be developed on recovery endpoints and the management plan—before anyone feels the need to sue. This approach, currently being used by EPA's Office of Water (the Watershed Protection Approach), is essential where management for recovery is a key goal because the recovery plan will need to accommodate many viewpoints over the long term. Moreover, once a consensus is established, coalitions must be formed to protect the plan from future interventions.

Determining who should be involved in this collaborative process involves answering the following questions:

- Who makes policy?
- Who defines policy options?
- Who chooses options?
- Who sets management goals?
- Who implements goals?
- Who pays and in what currency?
- Who benefits and in what ways?

Adequately answering these questions requires that the risk assessment team confer with risk managers and the public.

Once interest groups are identified and their viewpoints are represented, the risk assessment team can begin resolving issues concerning, for example:

- The starting point and where it fits within appropriate spatial and temporal scales.
- The characteristics or indicators and possible effects of disturbance.
- The goals for recovery and ecological endpoints (e.g., What ecosystem would be desirable following recovery? Is the current ecosystem the desirable system, or is some historical ecosystem sought, or is an entirely new ecosystem most desirable?).

Such issues must be evaluated in the context of natural and human-caused disturbance. While some anthropogenic disturbances (e.g., erosion, flooding, fire, pest outbreaks) have natural analogs that can be instructive as models for recovery, others (e.g., introduction of exotic chemicals, massive destruction of soils or sediments) may have no natural analogs and thus provide models for recovery that are speculative at best. To thoroughly evaluate these questions and choose applicable models, the risk assessment team must have access to the best scientific knowledge available.

2.2. What Ancillary Disturbances or Changes Are Occurring?

The risk assessment team, obviously, will focus on the disturbance that is the subject of the analysis because recovery from that stress is the reason for the analysis. The danger of a single-minded adherence to a too-narrow approach is that it can lead to the age-old scientific mindset that advises "hold all other variables constant." Unfortunately, the real world rarely holds all variables but one constant. Thus, when risk analyses are performed to determine the effects of specific stresses, the analyses also must take into account other changes occurring in the ecosystem that have little or nothing to do with focal concern. These can include climatic influences, biological invasions, chemical alterations, recreational pressures, and commercial impacts. The risk assessment team must not ignore these ancillary changes because they can significantly influence the desired recovery.

3. THE CONCEPTUAL MODEL: BASIC FACTORS INFLUENCING ECOLOGICAL RECOVERY

The risk assessment team should make every attempt to develop a formal conceptual model of the ecosystem at issue. The model should explicitly account for all pertinent components that can influence recovery, with linkages between components clearly described, and address driving variables and their temporal and spatial characteristics. Once the component-level conceptual model is established, submodels should be developed that emphasize internal controls and feedbacks within each component.

Developing a conceptual modeling approach calls for analysis of nine basic ecosystem components: weather, water, soil properties, assemblages of organisms, energy, economic viability, cultural and community viability, organizational viability, and political, policy-making, legal, and regulatory influences. All are interactive and interdependent, and none should be factored out of a recovery analysis without thoughtful consideration. Although the components are not of equal importance in all systems, many scientists, educators, land managers, policy-makers, and members of the public tend to overlook their interactions, failing to recognize that managing lands and water to ensure desirable recovery depends on their integration.

In this section we present a brief and general discussion of the individual components that must be considered in risk assessment regarding recovery.

3.1. Physical and Biological Factors

3.1.1. *Weather and Climate*

Weather and climate are particularly important factors in ecosystem modeling because they govern recovery. The growing consensus that global climate patterns are being altered by human activity poses a special concern to terrestrial ecosystem functioning and management. The critical aspects of weather and climate in regard to ecosystem functioning include precipitation amounts, pattern, and seasonality.

Similarly, daytime and nighttime temperature norms and extremes, seasonal temperatures, and the length of the growing season regulate ecosystems and determine their biological productivity. Additionally, the frequency and severity of extreme events (e.g., hail, frosts, flood) are important. If any of these parameters changes dramatically, it is almost certain that system functions and components will also change.

Less well understood but potentially important weather and climate characteristics include the atmosphere's chemistry (e.g., trace gases, carbon dioxide, particulates) and radiation transmissivity (i.e., ultraviolet radiation, quality, and intensity) as well as cloud cover. The direct and indirect effects on ecosystem composition and functioning of these characteristics are the subject of intense research and debate.

3.1.2. *Water*

Because water is a critical element of all ecosystems, particularly in the arid and semiarid regions of the world, it is profoundly important in determining ecosystem management success. Changes in the amount and quality of water, often related to its seasonal availability and its distribution, can completely alter the functioning of ecosystems. Because water is intimately related to both climate and human activity, it demands special attention in any recovery analysis.

Humans demand water for domestic, industrial, and agricultural uses and satisfy this demand by impounding and diverting surface water, regulating stream flows, pumping ground water, and designing sophisticated water allocation and distribution schemes. In some areas, water has come to be the chief point of tension between agricultural and urban sectors. The effects of chemical use and soil erosion, for instance, take on greater significance when translated into "off-farm" effects related to surface and ground water. Indeed, in many areas, urban and recreational needs have come to outweigh agricultural uses of water.

3.1.3. *Soil Properties*

Many soil properties have a profound influence on the physical and biological integrity and the sustainability of ecosystems. Thus, analysis of the potential for recovery of a disturbed ecosystem is critically dependent on the response and behavior of the system's soils. Prominent characteristics of soil that should be considered in developing the conceptual model are texture, structure, and volume; organic matter and forms; nutrient availability; erodibility and sedimentation; the functioning of soil organisms; and the degree of contamination from pollution. The nature of land management practices used at a site also can profoundly influence soil properties.

Knowing how much change soil properties can tolerate without jeopardizing the functioning of the entire ecosystem is often important information. Indeed, if there is one inviolate rule regarding sustainability of ecosystems, it is that basic soil resources must be safeguarded to maintain desirable terrestrial ecosystems.

3.1.4. *Assemblages of Organisms*

Defining the nature of assemblages of organisms involves identifying desirable organisms for the recovering ecosystem as well as characteristics required for the assemblage's maintenance, both above and below ground and in streams, rivers, lakes, and reservoirs. For example, the risk assessment team must ask, Is the ecosystem currently close to its "natural" or "native" state and can that state be maintained with minimal human intervention? If the desired ecosystem is not based on native species, will it be self-maintaining (i.e., will it perpetuate itself under current and future climate, water, and soil conditions)? Is the ecosystem capable of producing crops and supporting animals that meet current and potential future needs, and what kinds and degree of human intervention are necessary to ensure successful and profitable production? Are game or nongame species a desirable goal of ecosystem management? Is biological diversity an important issue in managing the ecosystem? What is the status and significance of weeds, pests, and diseases associated with the system? Is the current or proposed management scheme compatible with maintaining aesthetically valuable organisms and other environmental goals imposed by society?

Answers to such questions with reference to specific ecosystems are essential for wise management during recovery. Risk analysis, assessment of the impacts of stress and disturbance, and consideration of the prospects for recovery must be based on the best understanding of ecological principles as applied to the particular ecosystem. This understanding relies on the integration of knowledge concerning the ecosystem's physical and biological factors (addressed generally in section 4).

3.2. Economic, Social, and Organizational Factors

Establishing the goals and criteria for the desirable recovery and sustainability of an ecosystem is a social construct because human activity causes many disturbances and influences the subsequent recovery processes. Moreover, people assess and evaluate the state and progress of the effected ecosystem. Thus, forecasting the recovery of ecosystems from specific stress regimes presents an assessment challenge that calls for an integrated consideration of natural and social systems. This approach is especially relevant for agriculture, forestry, and fisheries systems, where the traditional paradigm has been to subvert natural processes with technology to achieve social goals. Creating, recreating, and approximating ecological processes ultimately requires a change in the social management, description, and assessment of ecosystems.

3.2.1. *Energy*

Energy is often both an input and output associated with ecosystems. Many managed ecosystems that are the subject of risk assessments produce timber and fuel wood, agricultural crops, or fish. Such ecosystems usually require machines, fertilizers, and/or pesticides for development and maintenance. The interrelationship of energy and agriculture, for example, is particularly important to our understanding of ecosystem management (CAST, 1992). Thus, the potential for recovery cannot be assessed without considering the availability, efficiency, and sustainability of energy.

3.2.2. *Economic Viability*

Most managed ecosystems are expected by society to produce valued commodities or products well into the future. Indeed, while an extensive body of literature on agricultural (e.g., Hazell and Norton, 1986; Ikerd, 1990), forestry, and fisheries economics (Costanza, 1991) addresses issues concerning the commercial sustainability of ecosystems, much less is known about the economics of recreation, aesthetics, and other nonmonetary values of ecosystems. The challenge for the risk assessment team is to account for the economic and nonmonetary value of the ecosystem and determine how the associated stress regimes will affect recovery. Further, the risk assessment team must evaluate important trends in recovery criteria and forecasts of future conditions.

Economics may not be a direct or principal issue in all cases, and ecosystem impacts on "off-site" environmental and social values will need to be more closely scrutinized in the future (Costanza, 1991). For example, the response of farmers and ranchers, foresters, recreational resource managers, and fisheries managers to climatic change may be influenced by subsidies that require ecosystem recovery and sustainability. Moreover, despite the importance of economic considerations, increasingly people are giving nonmonetary value importance in assessing the environments in which they live. Such priorities, which seem to have particular appeal in urban communities, can conflict with the priorities of people who depend on timber, fishing, and agriculture for their livelihood. If such aesthetic and spiritual values gain primacy, society will need to account for the incomes lost from hampering those enterprises.

3.2.3. *Cultural Influences*

Recovering ecosystems have a strong interplay with the human communities with which they are associated (Hart, 1991). Indeed, the success of any overall management practice intended to enhance recovery is dependent on the acceptance by the human communities associated with the ecosystem.

Thus, the risk assessment team must take into account the cultural setting, its relationship to existing ecosystems and stress regimes, and cultural trends.

3.2.4. *Organizational Viability*

Recovery analysis must take into account organizational bodies involved in research, management, and policy-setting concerning the ecosystem at issue. An organization's funding priorities and stability provide strong indications of how it integrates its mission and mandate with its goals, objectives, and political orientation. For example, an organization that accepts the concept of sustaining ecological recovery as part of its mission must set its priorities to manage for that purpose. Because sustainability is a concept with long-term implications, funding and human resources must be available on a stable basis. As a result, the risk assessment team must address such questions as, Is the relevant organization sufficiently established such that long-term missions can be accomplished, or is recovery likely to be hindered by future changes in the economic, social, or political climate?

All organizations develop their own peculiar culture, history, myths, and dogma. Public attitudes about the role of an organization are shaped both by how the organization is viewed in the social context and how it views itself. Thus, to implement policies and management practices that encourage sustainable ecosystem recovery, an organization must have suitable organizational dynamics and political viability. Understanding the functioning of key organizations and governmental bodies, as well as the interactions of organizations with overlapping jurisdictions, is essential in recovery analysis. Among the main organizational characteristics to be evaluated are mission or mandate; goals and objectives; human resource structure; funding priorities and stability; and ability to cope with change. Because organizational goals usually reflect short time horizons, a major challenge in recovery analysis involves determining whether the organizational goals and objectives aid or impede ecological recovery.

3.2.5. *Politics, Policy, Laws, and Regulations*

Politics, governmental policies, laws, and regulations are clearly related to the effectiveness of any ecosystem recovery scheme. Government policies and regulations provide a broad and important context for managing ecosystems in general and managing for recovery specifically. The risk assessment team's analysis must include an evaluation of the efficacy of implementing regulations that are intended to ensure a reasonable chance of ecosystem recovery. Questions that must be raised as an explicit part of an analysis include: Are there significant political, financial, or social impediments

to implementing policies, regulations, or management goals? Are there significant natural or human-caused changes in the biophysical controls of the system that might interfere with the recovery of the ecosystem? Similarly, are there significant economic, social, or institutional factors that might interfere with recovery? Once such questions have been addressed, different policy scenarios should be developed and factored into the recovery analysis with an eye to future developments and interactions.

4. THE CONCEPTUAL MODEL: ECOSYSTEM STABILITY AND FLEXIBILITY

4.1. Disturbance-Related Concepts and Terms

4.1.1. *Stability*

An ecosystem's stability characteristics constitute the larger conceptual framework for recovery. There is some confusion, however, about the definition of stability among ecologists of various disciplines. For example, Holling (1973) approached stability largely from the population perspective, defining *resilience* as persistence and *stability* as return to equilibrium after disturbance. These definitions are at variance with the understanding of stability generally adopted in the ecological literature since a definition was developed by Webster et al. (1975).

As used in this discussion, stability is defined as the nature of an ecosystem's response to a small, disturbance-related displacement from its normal trajectory (Webster et al., 1975). Stability consists of two components: resistance, which concerns the capacity of the system to avoid deflection in state despite a disturbance, and resilience, which is the ability of the system to regain its initial state once deflected.

4.1.2. *Disturbance and Stress*

Disturbance is generally accepted to be a discrete perturbation resulting in the loss of resident organisms and providing an opportunity for nonnative species to become established (Sousa, 1984). As such, disturbance changes the ratio of organisms to resources in a given population, community, or ecosystem (Pickett and White, 1985). Disturbance can be caused by a host of agents, either biological or physical. Thus wind, flooding, fire, and drought are considered disturbances, as are insect (pest) outbreaks, bioturbation, and localized, intense predation or grazing. Pickett et al. (1989) further define disturbance as an agent that is external to the system of interest; thus, an agent of disturbance from within an ecosystem (e.g., a stream) is a disturbance at the scale of the subsystem that experiences the effect (e.g., algae on a rock), but is not a disturbance of the system as a whole. Similarly, a significant spate in a stream or river is a disturbance at the temporal scale in regard to the weeks or months over which effects are manifested. Further, over the longer term (years), floods are not considered disturbances; rather, the absence of floods may be regarded as a disturbance.

Because disturbances are complex, they must be thoroughly described in any ecological analysis. Pickett and White (1985) and Sousa (1984) provide lists of disturbance attributes to consider when performing an analysis that include areal extent, magnitude, frequency, return time, and the predictability of disturbance events. Disturbance is a variable influence requiring careful description

and recognition that ecosystem response in terms of resistance and resilience will depend on the nature of the disturbance. Unfortunately, too little is known about ecosystem response to variable disturbances, to sequential events (i.e., disturbance regimes), and to simultaneous disturbances. Because disturbance seldom occurs as a discrete, suprathreshold event, it is difficult to adequately predict disturbance effects in advance of the event. Thus, more research is needed in this area.

Resh et al. (1988) define disturbance as an unpredictable event (i.e., outside the range of events normally experienced by an ecosystem). This definition distinguishes events that are part of the cycle of an ecosystem from those outside the system's trajectory, leaving undefined various events that cause devastating mortality but that may occur predictably (e.g., flood and drought, fire, pest outbreaks). Rykiel (1985) provides a mechanism for avoiding this problem by defining disturbance as an agent apart from its effect (i.e., a potential cause) and perturbation (and stress) as the effects of the event. Rykiel differentiates stress from perturbation by its level of action, measuring stress as a physiological or functional change in organisms and perturbation as the influence on ecological components or processes.

Barrett et al. (1976) define stress as a perturbation that is either foreign or natural to a system but applied at an excessive level. Kelly and Harwell (1990) define perturbation, stress, and disturbance synonymously. Because stress is defined by Niemi et al. (1990) as a component of a disturbance syndrome associated with an identifiable agent (the stressor), stress and disturbance could be understood as being synonymous terms when a single stressor is involved. According to these definitions, disturbance and stress move the ecosystem—or a component of it—outside of the range of states it "normally" experiences. While it may be clear that such stressors as heavy metals and toxic organic compounds are outside the ecosystem's normal experience, making such a determination about natural events such as flood and fire requires imposition of an arbitrary time scale. Characterizing the duration of disturbance presents a conceptual difficulty that was addressed by Bender et al. (1984) by differentiating between pulse disturbances and press disturbances, which involve a longer time frame. Thus, a flash flood may be considered a pulse disturbance, while drought is thought of as a press disturbance (Stanley and Fisher, 1991).

We propose that disturbance be defined as an event that potentially displaces the system of interest (e.g., population, community, ecosystem) from its normal trajectory (i.e., the one it otherwise would have taken). This definition focuses on the event rather than the effect, since events that are disturbances in one type of system may not be significant in another and a range of effects may result from the same event experienced by a range of ecosystems, communities, or populations.

Thus, the first step in the analysis of the ecological risk associated with any disturbance is to fully describe the disturbance event (e.g., Pickett and White, 1985) and its probable effects. This description must include consideration of disturbance attributes that in the past had been dealt with in an arbitrary and often conflicting manner, such as perturbation, pulse, press, stress, and predictability. Not only does this approach make ecological sense, it also is consistent with ecological risk assessment schemes that include an analysis of exposure and biological effects of anthropogenic stressors.

4.1.3. *Resistance*

Resistance is measured as the inertia of state variables across a disturbance. In essence, assessing resistance involves a before-and-after analysis of the effect of the event. Units can reflect absolute or percent change; several variables should be measured, depending on the assessment endpoints (e.g., population size and the structure of the dominant and keystone species, exemplary taxa representing the resident ecological guilds, and species of special interest such as endangered, commercial, or sport species). At the community level, species richness, equitability, spatial pattern, and niche partitioning should be considered. Primary productivity, photosynthesis:respiration (P:R) ratio, ecosystem-level nutrient retention, and decomposition rates could be included when considering the response of an entire ecosystem.

4.1.4. *Resilience/Recovery*

Resilience refers to the capacity of the system to regain its previous state once the disturbance is removed. Measurement endpoints should be assessed with the same variables used to describe resistance, even though in resilience analysis several variant expressions are available. For instance, the time required to reach the predisturbance state may be used or, in the case of asymptotic recovery trajectories, half-saturation times may be used. Also, the rate of change in the measurement endpoint may be used to reflect resilience, indicating the rate of initial change without assuming that the "before" state will be reached (Grimm and Fisher, 1989).

Because populations, communities, and ecosystems are constantly changing in response both to the most recent disturbance and seasonal and long-term climatic pressures, risk assessment presents significant complexity. Often it is unrealistic to expect recovery to the predisturbance state, rather than to some point on the predisturbance trajectory. For example, a grassland burned in spring will likely be populated in summer by predictable insect species. Similarly, stream insect communities of

the Southwest devastated by flash floods recover to trajectories determined by underlying seasonal change (Boulton et al., 1992a). Thus, selection of measurement endpoints for recovery should be realistically selected after considering (1) the predisturbance state at the study site, (2) nearby undisturbed sites, and (3) the literature describing recovery patterns in replicate ecosystems of the type in question.

Resilience, recovery, succession, and ecosystem development are somewhat synonymous and often are used interchangeably to describe postdisturbance change. Resilience is a neutral value but was used by Holling (1973) to refer to what most ecologists now call resistance; thus, some confusion may persist. *Recovery* implies a state of response to stress from disturbance. *Succession*, a classic concept in ecology, describes the sequence and causes of change in populations and communities following a disturbance; however, because the conceptual bounds and implications of succession are not agreed upon by the broader ecological community, this term is best avoided (Fisher, 1990). *Ecosystem development* is a term used by Odum (1969) to describe succession in ecosystems such that a variety of ecosystem-level properties of dubious value and uncertain origin are incorporated; the concept is historically related to the Clementsian supraorganism concept (Clements, 1916), which has been largely discredited. Odum considers succession in ecosystems to follow an ordered, predictable process characterized by regular changes in several holistic properties over time. Similar models were presented by Margalef (1963, 1968), who described succession in terms of changes in pigment ratios, photosynthesis:biomass (P:B) ratio, P:R ratio, and information (or its inverse, entropy). Margalef's uniquely ecosystem-oriented view implies a certain degree of control over the process of succession described in terms of collective properties (e.g., P/R, P/B, tightness of nutrient cycling, entropy). This concept is useful because it focuses on ecosystem services, but it is tangential to traditional biologic succession, which focuses on organisms and communities. Moreover, this supraorganism view is held in some disfavor by evolutionary ecologists because it presupposes a cybernetic function of ecosystems for which there is little evidence but no shortage of debate (Engelberg and Boyarsky, 1979; Knight and Swaney, 1981; McNaughton and Coughenour, 1981). Many of the ecosystem-level trends hypothesized by Odum were drawn from and influenced by studies of autotrophic ecosystems such as forests and aquarium-style microcosms in autotrophic succession. Unfortunately, many ecosystems are heterotrophic, depending on energy derived outside their boundaries, and deviate from the developmental trends proposed by Odum (see Fisher et al., 1982).

4.2. Factors Influencing the Recovery of Ecosystems

In this section, recovery specifically refers to postdisturbance changes in biological systems. This use of the term is not intended to imply that systems in a state of recovery are unusual, undesirable, or ecologically deficient apart from such a judgment established in reference to assessment endpoints. If the problem formulation step in ecological risk assessment identifies and assigns values to various assessment endpoints (e.g., species diversity, primary productivity, population size), then these endpoints will serve to differentiate ecosystem states in terms of human-assigned values. Such values are imposed by the assessor and are not otherwise ecologically inherent in the disturbed system.

4.2.1. *Resistance*

In general, resistance and resilience are negatively correlated in ecosystems (Sousa, 1984; Webster et al., 1983). Thus, ecosystems with a high capacity to resist disturbance (or stress) tend to have a limited ability to recover from disturbance events, and vice versa. As a result, it would be incorrect to assume that a badly damaged ecosystem would experience a slow or difficult recovery period; indeed, the opposite is more likely to be true. Also, an ecosystem that is only slightly altered by a disturbance may recover quite slowly. In assessing recovery potential, it is important to understand the overall stability characteristics of an ecosystem to make realistic predictions about the capacity for recovery.

Resistance is a particularly important component of conceptual modeling because it establishes the initial conditions of the recovery process. The starting point for recovery may characterize an ecosystem as one that has been changed considerably by a short-lived disturbance event (e.g., by a fire, a hurricane, a chemical spill) or one that has been subjected to a prolonged period of moderate stress. In either case, the most important predictor of recovery capacity is the system's state once the stress is removed so that recovery can begin. This state can be described in individual, population, community, or ecosystem terms and may be weighted for assessment endpoints selected by the risk assessment team.

4.2.1.1. Mechanisms of Resistance

Physical factors. Physical factors are an important component of resistance in any ecosystem. For example, flood effects can be moderated by streambed slope, extent of lateral wetlands, floodplain size, and the position and extent of levees. Flood effects also depend on substrate size and distribution. For example, a boulder-cobble bottom streambed will resist scouring to a greater degree

than a sand bottom streambed, which experiences a longer period of disturbance because the streambed is mobile for a longer period during flood recession. Although clay substrates exhibit greater resistance than sandy substrates, once entrained they settle extremely slowly (Morisawa, 1968) and the disturbance event is prolonged (Wood et al., 1992).

Resistance of lake ecosystems to disturbance is a function of the relationship between depth and area (i.e., the hypsographic curve), circulation patterns, and degree of stratification. These properties influence exposure of the biota to toxic substances that enter through hydrologic pathways. The interaction of the hypsographic and stratification characteristics determines the extent to which toxic materials come in contact with the benthic community. Considering transparency, this interaction also determines the extent to which autotrophic and heterotrophic ecosystem components are influenced by the event. For example, a lake in which a large volume of water overlies—and therefore may interact with—the littoral zone may respond differently than one in which contact with the littoral zone is limited (e.g., by a steep shoreline).

Although terrestrial ecosystem resistance is largely a function of the biotic state, physical factors play a role as well. For example, fire propagation is a function of the moisture content of fuels, but it is also related to weather (e.g., precipitation, relative humidity). Similarly, landscape patchiness, which is a physical property, may influence disturbance size and shape (e.g., as a function of patch size, percolation, and the size, shape, and position of firebreaks such as roads).

Successional age. Successional age (i.e., time elapsed since the last disturbance) influences the extent to which subsequent disturbances change target ecosystems. For example, early successional ecosystems may not be significantly affected by disturbance events that occur in rapid succession for the obvious reason that little remains to be further damaged. In some streams, however, recolonization occurs rapidly and pioneer communities quickly establish themselves. Thus, disturbance events separated by weeks, or even days, can add substantially to cumulative damage. Successive floods in alluvial streams may add to disturbance effects since the saturation of previously dry channel sediments can result in increased runoff compared with the first disturbance (i.e., lower resistance is associated with the second event). While this effect can extend throughout a watershed in mesic regions, it is largely confined to channels in arid lands that are characterized by minimal soil abstraction from rainfall and sheetflow runoff patterns (Fisher and Grimm, 1985; Fogel and Duckstein, 1970). In forests, however, because fuel buildup is a slow process that depends on the

mortality of large trees or tree parts, recently burned forests are somewhat immune to disturbance by fire since they exhibit higher resistance as a function of their disturbance history.

Resistance of riparian vegetation in stream channels is also a function of successional age. Although early seedling stages show minimal resistance to annual floods, large riparian trees tend to be highly resistant. Baker (1990) demonstrated that recruitment of cottonwood seedlings is associated with hydrologic events recurring every 4 years; however, conditions resulting in stands of saplings that are the same age occur only every 11 years (i.e., the seedling stage vulnerable to floods of a magnitude that has minimal effect on saplings). When this 11-year opportunity arises, resistance of the riparian component of such ecosystems increases markedly. A similar situation exists with macrophytes in streams, where algal communities showing minimal resistance to spates typify the first 2 to 3 years of postflood succession in sandy rivers but eventually give way to macrophytes that are more firmly rooted in the stream substrate. Resistance of the macrophyte community is substantially higher than that of the early, algal phase (Wood et al., 1992).

Biogeochemical mitigation. When the affected ecosystem is described in terms of intact biogeochemical cycling, resistance to disturbance is assessed in terms of the extent to which these cycles are altered by a particular disturbance event. In ecosystems where nutrient cycling involves a spatial separation of processes, certain components may be differentially influenced by stress. Underground processes in forests, for instance, may not be significantly affected by fire and hyporheic processes may be immune to flooding. Similarly, although nitrification occurs in stream sediments and vertically transported nitrate supports the high rates of primary production of benthic plants (Valett, 1991), floods remove the benthic components of this cycle in such ecosystems. Since floodwater is high in both nitrate and ammonium, however, a flood can enhance rates of nitrification below the sediment surface. Thus, assessment of ecosystem resistance must take spatial aspects of system function into account.

Refugia. Refugia are extremely important in assessing the immediate and prolonged effect of disturbance. Indeed, any ecological assessment must be built on an understanding of the type, distribution, and capacity of refugia (Sedell et al., 1990) because their effectiveness determines the extent to which an ecosystem is altered by stress and concerns the starting point for subsequent recovery.

Refugia effectiveness is specific to the system under consideration and the organisms involved. For example, for spates in streams and rivers, substrate materials may provide a refuge for macroinvertebrates, insulating the community from all but the largest events, as is the case in Rocky Mountain and eastern U.S. trout streams (Peckarsky, 1986; Skinner and Arnold, 1988). In sand bottom streambeds, however, sediments provide no refuge for surface-dwelling macroinvertebrates; instead, refugia are found in the airspace of the riparian corridor (Gray and Fisher, 1981), in backwaters, in nearby undisturbed tributary streams, or in unflooded upstream reaches. The effectiveness of such refugia will determine not only the species and abundance of survivors, but the pathways of recolonization and the nature of pioneer communities. Interestingly, however, hyporheic organisms show relatively high resistance to floods and probably occupy refugia deep in sandy alluvial deposits (e.g., desert streams [Boulton et al., 1992b]) or in debris accumulations associated with the stream surface (e.g., streams in Virginia [Palmer, 1990]).

The concept of refugia is based on the understanding that disturbance effects are not uniform in space. Thus, it is necessary to understand the mode of disturbance to properly assess resistance and the potential for an ecosystem's recovery.

Individual adaptations. An array of adaptations enables individual organisms to resist disturbance as a result of natural selection operating over long periods. These selective forces are associated with the nature of the disturbance and are likely to be well developed where the type and nature of disturbance events have been predictable over an evolutionary period. Adaptations can involve morphological, physiological, or behavioral features of the organisms. For example, jack pine (*Pinus banksiana*) produces a cone that opens at the high temperatures associated with forest fires, utilizing a mechanism of resistance at the level of the genome as a facilitator of recovery. Similarly, many invertebrates of temporary ponds possess resting stages that facilitate survival during long dry periods. Mollusks of the coastal, rocky intertidal zone attach themselves firmly to substrate by byssal threads so they can maintain their position during storms and resist assaults by debris (e.g., logs) entrained in the crashing surf; during low tide, mollusks close their shells to avoid desiccation, a response that also may be used to avoid the effects of toxic chemical spills. Moreover, fish of the southwestern deserts of the United States have been shown to respond behaviorally to impending flash floods, swimming laterally away from the stream center and the full force of the disturbance. Interestingly, *Poeciliopsis occidentalis*, a native to the Southwest and a species with evolutionary

experience with floods, exhibits this behavior, while the closely related *Gambusia affinis*, from the hydrologically benign Mississippi drainage, does not (Meffe, 1984).

4.2.1.2. Limitations to Resistance

Because resistance is an essential component of stability, it greatly influences recovery and must be carefully considered in any assessment of ecological risk. An important consideration is that several factors may limit "natural" resistance in many ecosystems. In particular, since attributes associated with resistance often are keyed to the organism level of an ecosystem rather than to populations and communities, it is more important for genetic traits to be passed on despite the disturbance event than for ecosystem functioning to remain intact. Nonetheless, adequate capacity for ecosystem resistance depends on the maintenance of reasonably intact, structurally complex functioning and a full set of refugia. Often, simplified systems are vulnerable to disturbance because they do not possess the array of resistance features available in "natural" ecosystems. Also, communities or ecosystems that have undergone species substitution may lose their capacity to resist the stress of disturbance (Meffe, 1984).

Resistance to disturbance is in large part a characteristic of organisms that have been shaped by natural selection in an environment characterized by a limited set of disturbance types and magnitudes, regardless of severity. In ecosystems with such organisms, subtle changes in the intensity, timing, or magnitude of disturbance beyond the range experienced by resident species can have devastating effects. Additionally, such ecosystems can be eradicated by novel disturbances with which the community has no evolutionary experience (e.g., radioactivity, organic solvents, outbreaks of exotic pests).

Unfortunately, our ecological understanding of the complex of factors that confer resistance characteristics—at both the ecosystem and organism levels—is lacking, presenting a challenge for ecological risk assessment until more research is conducted.

4.2.2. *Disturbance Type*

4.2.2.1. Various Disturbances

Many types of disturbances occur in the wide variety of ecosystems that exist, since disturbance is the rule rather than the exception. Indeed, most ecosystems consist of communities of individuals that are perturbation dependent. Yet alteration in disturbance type, magnitude, or frequency can cause greater changes than those associated with the established disturbance regime.

Rain and flooding. Flood disturbance is typical for many stream ecosystems, especially in arid regions (Fisher et al., 1982; Siegfried and Knight, 1977), and even essential for the proper ecological functioning of large flood-plain rivers (Sparks et al., 1990). Similarly, although riparian ecosystems may experience substantial mortality during extensive flooding, the continued existence of such complex systems depends on periodic disturbance (Campbell and Green, 1968). In contrast, excessive rainfall may favor the growth of annuals in terrestrial ecosystems, which then increases their susceptibility to fire (Vogl, 1980).

Wind. Hurricanes and cyclones represent formidable agents of disturbance, especially in such areas as south Florida and the Gulf Coast, but often far beyond these areas as well. In 1938, for instance, hurricane damage to hardwood forests in New England was extensive. Although such disturbance events can be damaging to habitat and wildlife alike, diversity is often maintained by, for example, the rejuvenation of dunes, the opening of new habitat, the dispersal of seeds, and the moderation of competitive exclusion.

Fire. Fire is a catastrophe in forests and grasslands that can significantly reduce biomass and initiate lengthy successional recovery sequences. Nonetheless, many ecosystems are fire-dependent. While some plants resist fire by relying on thick bark or underground meristems, others (e.g., the Pin cherry) depend on fire and sprout readily from seeds in fire-damaged areas (Marks, 1974). Fire intensity ranges widely from rapidly moving ground fires with minimal effect on canopy trees to crown fires, which are devastating to mature vegetation. On a landscape scale, fires maintain a patchy horizontal structure that supports a diverse, larger community consisting of several local assemblages in various states of succession.

Erosion. Catastrophic erosion is associated with precipitation events and typically occurs on steep slopes covered with a mantle of unconsolidated soils. Soil slumps, landslides, and debris flows (Swanson et al., 1987) are relatively rare events that alter geomorphology, soil dynamics, and, in the case of stream channels, fluvial geomorphology (Lamberti et al., 1991). For example, the extensive arroyos cut into the desert landscape of the southwestern United States in the late 1800s were caused by a combination of overgrazing and vegetation changes associated with climate shifts (Hastings and Turner, 1965). This alteration in the landscape led to lowered water tables and the loss of riparian plants, as well as changes in the geomorphology of stream channels and the draining of cienegas (i.e.,

wetlands), which had been extensive in the region. As with debris flows in the Pacific Northwest, recovery from arroyo-cutting episodes is slow (Hendrickson and Minckley, 1984).

Drought. Periods of low water availability represent a stress for both terrestrial and aquatic ecosystems. Although droughts tend to be prolonged rather than recurrent disturbances, both their onset and termination often are easily recognized. In lakes and ponds, drought influences the residence time of water as well as dissolved and suspended substances and, in extreme cases, can lead to salinization of output water, which occurs largely by evaporation rather than seepage. Generally, droughts have a more severe effect on reservoirs, which are usually impoundments of water from large rivers, than natural lakes because turnover time (i.e., response time) is short.

Streams are influenced by droughts in several ways. For example, in permanent streams limited flow can restrict available habitat, ultimately yielding a discontinuous series of stream reaches. In desert streams, droughts tend to have a greater effect on habitat loss than flash flooding in terms of mortality and productivity (Stanley, 1993).

Recovery after a drought can be impaired if populations of potential colonizers of the newly inundated stream channel are reduced. Depending on the pattern of drought, available colonizers may relocate in significant numbers from the main channel of a stream to tributaries of local springs. Such shifts of colonizers render prediction of recovery patterns uncertain.

Since fish show little or no resistance to dewatering associated with drought in bodies of water in North America, such disturbance events can cause high mortality. Thus, in most cases, recovery is dependent on a continuous aquatic connection with a colonization source—usually downstream areas that remain permanently wetted. Biomass recovery is usually rapid following rewatering; however, the species composition of the new community is likely to recover more slowly (Larimore et al., 1959).

Riparian areas and wetlands can be lost when a receding water table drops below the rooting zone. Reestablishment of vegetation in such areas requires restoration of the ground-water table to its previous level during the time of year when seedlings are established (Stromberg and Patten, 1990).

While droughts are among the most common disturbance events in nature, little is known about their effect on streams and rivers because fluvial ecologists are not adept at dealing with ecosystems that expand and contract. Needed research on drought effects at a variety of spatial scales is likely to be increasingly important, given impending global climate changes and accelerated regional desertification.

Animal-caused perturbations. Although biotic effects are endogenously generated, disturbance attributable to animal activities can be quite pronounced in ecosystems. In aquatic systems, grazing by invertebrates or fish can markedly influence the abundance and structure of the algal community (Power et al., 1985), an occurrence often related to predation transmitted through complex food chains. For example, abundant predators reduce grazer populations, releasing algae from grazer control and resulting in luxuriant growths (Power, 1990).

In addition to animals, pathogens represent endogenous agents of disturbance. In an Arizona stream, epidemics of a bacterial pathogen infecting diatoms can cause a loss of biomass and productivity, similar to that caused by a flash flood, without directly influencing the rest of the trophic structure of the system (Peterson et al., 1993).

In terrestrial systems, overgrazing by cattle can significantly affect plant community structure and productivity (Ferguson and Ferguson, 1983). The concentrated browsing of elk and moose, however, maintains openings in the boreal forest. In the grasslands of the western United States, openings are similarly maintained by the grazing of large mammals. Similarly, sea and marsh birds may kill vegetation in nesting areas by depositing large quantities of guano (Vogl, 1980). Also, acacia-associated ants have been found to clear vegetation and debris from below host trees. Although this is itself a disturbance, it reduces the probability of disturbance from fire (Sousa, 1984).

4.2.2.2. Natural Versus Anthropogenic Stress

Anthropogenic disturbance can differ from natural disturbance in several ways. It can:

- Increase or decrease disturbance intensity;
- Alter disturbance regimes, frequency, or timing;
- Be induced by unique stressors (e.g., toxic chemicals, exotic organisms);
- Be applied in a unique spatial pattern; or
- Be superimposed on ecosystems that are already stressed by an array of natural agents.

In short, anthropogenic stress tends to influence organisms in a manner that is well outside their evolutionary experience, often altering communities and ecosystems in dramatic and unpredictable ways.

Human activity also can ameliorate natural disturbances by influencing their frequency, magnitude, or expansiveness. Thus, fire-control policy can result in less-frequent fires, and flood-control efforts can minimize disturbance by spates. In contrast, hydropower generation can produce intertidal zones in freshwaters that have a periodicity to which organisms are not adapted (Fisher and

LaVoy, 1972). Similarly, the clear-cutting of forests can remove ecosystem nutrients along with forest products and disturb soils. This combination of anthropogenic effects has no natural analog. Superimposed on these ecosystems are community changes resulting either from altered disturbance regimes or introduced, exotic species.

In addition to altering natural disturbance regimes, human activity can introduce stressors that are novel in terms of the evolutionary history of the resident organisms (e.g., oil spills, organic solvents, radioactive substances, heavy metals). Indeed, anthropogenic stresses trigger responses from the organism to the ecosystem level of organization that are difficult to predict from research on natural systems. Once stresses are alleviated, recovery can be better assessed from general studies that have been modified to account for an altered starting point. Thus, there is no reason to think that a flood-damaged stream and one influenced by a petrochemical spill will not recover by similar mechanisms, assuming colonization sources remain intact. On the other hand, recovery of large, fire-damaged areas may be slower than recovery for smaller areas as a result of greater colonization distances.

4.2.3. *The Spatial and Temporal Nature of Disturbance*

Because recovery is likely to be sensitive to the spatial nature of disturbance and the time (at several scales) at which it occurs, ecological risk assessment must include an accurate description of the disturbance event relative to space and time. Although space is relatively easy to describe as the area affected (Sousa, 1984), a more useful description must address the area's relationship to geographic, topographic, environmental, and community gradients (White and Pickett, 1985). This more comprehensive approach to defining the spatial nature of a disturbance is critical if the stressor does not encompass the entire ecosystem (e.g., the disturbance may differentially affect sites that are important as refugia). Although all disturbances affecting a given area should be described separately, it also may be appropriate in some cases to describe the spatial nature of disturbance in terms of its recurrence within a particular time frame.

Recovery can be affected by altered colonization distances, especially in cases where refugia are destroyed. Thus, the availability of colonizers or their propagules may depend on the spatial extent of the disturbance. For this reason, the affected area should be expressed as a percentage of the ecosystem's total area.

The timing of a disturbance can significantly influence both resistance and resilience and should be considered at several scales (e.g., time of day, season, successional time). For instance, seasonal

phenology will determine differential exposure of ecosystem components to the disturbance event (e.g., spring fires will differentially affect spring herbs); also, time of day can be important when disturbances are recurrent. Similarly, in desert streams, where flash floods last only a few hours, many species of emergent insects in the aerial corridor above the stream, which represent the primary source of colonists, live less than a day as adults. In such environments, the adult populations of mayflies and chironomids may flourish as a result of their emergence from early evening through early morning hours, while late afternoon populations are likely to be greatly reduced by daytime mortality. Thus, late afternoon floods can have a more significant impact on these insect populations than floods that occur after dark when emergence is well under way and can influence the rate of postflood recovery (Jackson and Fisher, 1986). Also, because some mayfly species emerge only in early morning hours, their contribution to the colonization pool is increased if storms generate late-night flash floods.

A temporal description of disturbance should include a measure of the disturbance frequency and return interval as well as the rotation period (i.e., time required to disturb an area equivalent to the study area [White and Pickett, 1985]). Such measures gauge the availability and successional age of nearby systems that might contribute colonizers to the disturbed site along with facilitating assessment of the effect of sequential events on the system. For example, flash floods in rapid succession in a desert stream would have a greater effect on the development time of the dominant taxa than the same number of floods separated by several weeks.

4.2.4. *Disturbance Scale*

Along with contributing to the definition of a disturbance's spatial nature, scale effects are important in ecological risk assessment because assessment and measurement endpoints must be keyed to the same scales. For example, disturbances such as water drawdown and drought that alter ecosystem size tend to affect a large, heterogeneous area. Thus, for ecosystems subject to such disturbances, recovery of biological components (e.g., macroinvertebrates, fish) should be measured in a stratified manner to reflect this heterogeneity. Otherwise, biotic recovery at sites inundated with water, for example, will give a false measure of the recovery rate.

Disturbance scale also influences the variables that emerge as important for controlling a given recovery process. For example, postflood recovery of algal communities in streams is best explained in relation to the time that has elapsed—at the temporal scale of years—since the disturbance event; in the short term, however, rate of recovery is a function of nutrient availability (Grimm, 1994). Thus,

in postflood management, an assessment of the recovery potential must specify the scale to facilitate selecting between alternatives involving nutrient enhancement or flow regulation. Similarly, Duarte (1991) showed that macrophyte abundance (and presumably recovery potential) is a function of lake morphometry and transparency at a regional scale as well as a function of slope, exposure, and depth within a given lake. Moreover, at the scale of a given macrophyte bed, biotic interactions and sediment characteristics are the primary controlling factors. Thus, if endpoints involve macrophyte abundance in a particular littoral zone, it would be unproductive to assess recovery potential in terms of either transparency or exposure, since neither is relevant at this scale. Instead, the risk assessment team should focus on restoration of sediments and "normal" biotic interactions (i.e., factors that directly influence macrophyte biomass in a lake).

Even though natural disturbances may devastate communities on one scale, they may be necessary to maintain diversity and normal ecosystem functioning at the larger scale. For example, ecologists have come to appreciate the role of fire in the maintenance of large-scale ecosystems, accepting that fire prevention as a management approach intended to hasten "recovery" of disturbed patches can be counterproductive. Deciding when it is appropriate to control natural disturbances requires an understanding of landscape dynamics. In this area of study, aquatic systems in particular are poorly understood in terms of spatial heterogeneity and scale effects. Expansion of landscape tools, techniques, and concepts into the aquatic arena is needed (Fisher, 1994).

4.2.5. *Ecosystem Type*

Although ecosystem type has a significant effect on recovery, it is unlikely that any single recovery theory is sufficiently broad to embrace all types. Nonetheless, major differences among ecosystem types can be divided into four characteristics relevant to recovery:

- Status of existing standing biomass;
- Time required to regrow standing biomass;
- Mobility of the medium relative to vectors imposing agents of disturbance; and
- Connection with adjacent systems;

For example, terrestrial ecosystems disturbed by airborne agents will begin recovery immediately after the source of disturbance is eliminated. Lakes, however, may show prolonged effects, depending on hydrologic residence time and turnover of involved intrasystem storage pools (e.g., sediments). Streams and rivers lie somewhere in between these extremes because toxicants usually

are rapidly removed by flowing water. (Other examples of ecosystem-specific differences are discussed throughout this chapter.)

4.2.6. *Ecosystem Linkage*

Although recovery can result from either regrowth of ecosystem survivors or colonization from outside the system, it usually involves both mechanisms. Indeed, intact adjacent systems are often essential to significant, timely recovery following disturbance. Thus, assessment of ecological risk must be based on a knowledge of recovery mechanisms typifying the subject ecosystem—information that often must come from studies of similar systems.

Along with serving as a source of recolonizers, adjacent systems may be linked also in terms of long-term recovery. For example, when a stream as well as an adjacent forest are affected by a disturbance event, the recovery pattern may be altered. Thus, at New Hampshire's Hubbard Brook Experimental Forest, where deforestation was accompanied by increased runoff flow and transport of dissolved and suspended materials, algal communities in adjacent aquatic areas responded to increased nutrients with additional blooms. As a result, the rate of recovery was enhanced, but to a different, more autotrophic endpoint (Bormann et al., 1968). Webster et al. (1983) reported significantly delayed recovery of stream communities in a deforested watershed in North Carolina as a result of altered organic inputs from the adjacent terrestrial forest and suggested that stream recovery would depend on the much slower process of forest recovery. Indeed, it has been shown that food webs in forested streams are heavily dependent on organic inputs from adjacent forests (Fisher and Likens, 1973).

Thus, adequate risk assessment must be based on a thorough understanding of ecosystem linkages with respect to organisms, energy, and nutrients. Because strong interactions among ecosystems are the rule rather than the exception, recovery is likely to be strongly influenced by linkage.

4.2.7. *Biological Characteristics*

4.2.7.1. Individuals and Populations

The types of organisms in a community will influence the rate of recovery after disturbance because recovery depends on the regrowth of survivors and recolonization from refugia within the system or from nearby systems. The basic biological and life-history features of survivor and colonizer organisms, however, may be quite different. Although survivors demonstrate high resistance attributable to a variety of mechanisms, they are often larger, slower, and less vagile than

colonizers. Colonizers are early successional species that are generally vagile, fecund, prolific, rapidly growing, and often small (Vogl, 1980). Populations of colonizer species may experience high mortality during disturbance but then recover rapidly.

As with whole ecosystems, resistance and resilience show certain inverse correlations at the species level. This is particularly true for organisms in stream ecosystems that are subjected to flood disturbance. In such environments, small aquatic insects experience high mortality but recolonize rapidly; larger insects such as coleopterans and hemipterans are much more resistant to flooding and have substantially longer life cycles (Gray and Fisher, 1981). This principle is probably most applicable when disturbance is associated with some physical process to which body size confers a degree of immunity. It might not apply when disturbance is induced by anthropogenic agents such as toxins, for which dose response and susceptibility may be independent of body size. Samuels and Ladino (1983) found that a population of long-lived, slowly reproducing terns required 20 years to recover to predisturbance levels after an oil spill, while the more rapidly reproducing herring gull recovered in just 5 years.

Life history or "vital attributes" of populations can vary widely and can significantly influence the rate of recovery. Such attributes relate to the method of arrival at the disturbed site, the ability to grow in the postdisturbance environment, and the time needed to reach critical life stages (Noble and Slatyer, 1980). Recovery of a set of populations will depend on when such vital attributes are manifested relative to the likelihood of disturbance recurrence.

Life history characteristics also can vary spatially. For example, since attributes of fish species in a multispecies community may vary longitudinally, recolonizers that are small and developmentally fast tend to be more abundant in headwaters (Schlosser, 1990). Other factors being equal, the recovery of fish populations depends on the location of the disturbance in the drainage basin.

Moreover, the ability of an ecosystem's populations to interact can influence the successful recovery of the entire complex. For example, exotic grasses used to revegetate disturbed arctic sites were found to grow rapidly, but they were unable to establish permanently unless continuously fertilized. Although native grasses established themselves more slowly, they persisted without fertilization (Chapin and Chapin, 1980). In contrast, alders fix nitrogen and serve as nurse trees for slower-growing Douglas fir trees on sites where available nitrogen is scarce (Finegan, 1984). Thus, successful recovery of individual species populations is a function of physiology (e.g., nutrient uptake) as well as life history.

One of the greatest challenges in recovery management is to maintain conditions that encourage growth of the longest-lived species in a community. Because such species are often the slowest growing members of the community (Schlosser, 1990), management practices that focus on them are likely to protect other ecosystem populations as well (Gore et al., 1990).

4.2.7.2. Communities

Recovery of communities is assessed largely as a description of the temporal change in resident species. For some time, ecologists have argued about the "rules" that govern the assembly of communities on the temporal spectrum. Concerning terrestrial systems, there is little agreement as to whether assembly rules are universal; however, it is most likely that they are specific to the site (i.e., the community type) (Lawton, 1987).

Disturbance-generated change in abiotic conditions plays an important role in the initial assembly of communities. For example, clear-cut or herbicide-treated forests are initially wetter and rich with nitrogen compared with nearby reference sites (Bormann et al., 1968; Sollins et al., 1981). Such conditions hasten recovery and determine the success of potential colonizers, allowing assembly rules to be significantly influenced by biotic interactions (e.g., competition, predator-prey activity, pathogenesis, mutualism [Lawton, 1987]). These interactions can operate to varying degrees in individual ecosystems. Four models describe the typical sequence of community change: facilitation, tolerance, inhibition, and random colonization (Connell and Slatyer, 1977). The models differ primarily in the degree to which biological interactions influence the rate of recovery.

Three points relevant to environmental risk assessment emerge from a consideration of assembly rules:

- No one set of rules is applicable to ecosystems generally because similar ecosystems (e.g., forests, grasslands) may operate quite differently.
- Assembly rules may vary within a given recovery sequence and among components of a given community (e.g., early successional vegetation may demonstrate facilitation, while early successional birds may demonstrate development that is more consistent with inhibition models). Also, control may reverse in later successional communities at the same site.
- There is a strong stochastic component to community assembly such that stochastic early stages may "lock in" later stages. Indeed, alternate deterministic endpoints are dependent on random initial conditions (Lawton, 1987; Horn, 1981).

In ecological risk assessment, there is no adequate substitute for a thorough understanding of the ecology of the community type under consideration. Unfortunately, ecological research cannot always provide especially detailed information.

4.2.7.3. Ecosystems

Recovery at the level of the ecosystem involves reestablishment of biomass and ecosystem functioning measured as energy flow and nutrient cycling. For regrowth to occur, energy in the form of sunlight or organic matter must be available to support accretive processes. In terrestrial systems, sunlight is seldom limiting during initial stages of recovery. In lakes, however, increased turbidity or altered depth structure may change the distribution of light energy below the lake surface. Because in the aphotic zone of lakes, in underground environments of terrestrial ecosystems, and in streams and rivers, much of the available energy is allochthonous, restoration of normal inputs is a requisite to ecosystem recovery. For example, the recovery of streams disturbed by deforestation is limited until the forest watershed (especially the riparian zone) recovers to the point where allochthonous inputs are restored in quantity and quality (Webster et al., 1983; Golladay et al., 1989). In contrast, when disturbance of streams is caused by the release of caustic, acidic, or toxic materials, rapid recovery of ecosystem function can be expected (Cairns et al., 1971) once residual toxicity is removed because energy supplies remain largely unaffected. Release of sequestered toxics, however, can trigger a relapse (e.g., resuspension of river or stream sediments) (Suter, 1993).

Nutrient cycling at the ecosystem level can be measured as the extent to which the system retains nutrients (Odum, 1969). Since disturbance increases "leakiness," recovery can be measured in terms of this output:input ratio. Thus, since retention capacity is largely vested in biomass, full recovery of an ecosystem depends on regrowth of vegetation, which may take hundreds of years in temperate forests and grasslands (Likens et al., 1978). Recovery is often more complex than this, however, because late-stage successional ecosystems display a reduced net biomass increment and have the potential to become leaky, as has been demonstrated in forests (Vitousek and Reiners, 1975) and streams (Grimm, 1987), and logically applies to all ecosystems.

For nitrogen—and probably for many other elements—the processes involved in retention or export are especially complex. In terrestrial ecosystems, for example, nitrogen leakiness is a function of eight processes that vary regionally among 19 forest sites analyzed in the United States (Vitousek et al., 1979). The analysis found that while many processes result in nitrogen loss, vegetation regrowth was requisite in all cases to restore normal nutrient retention patterns.

At the scale of a heterogeneous landscape that consists of patches representing various stages of disturbance, retention of nutrients overall will reflect some mathematical average of the functioning of individual patches. A landscape comprising a mix of successional states will be more retentive than either a widely disturbed landscape or one uniformly covered with old-growth forest. Thus, recovery at this level involves maintenance of an optimal mix of landscape patches (i.e., states of recovery) rather than prevention of disturbance because stress may enhance the nutrient retention of the larger landscape rather than decrease it (DeAngelis et al., 1985).

Unfortunately, no routine assay has been developed to address changes in whole ecosystem processes such as P:R ratios and nutrient retention (Bartell et al., 1992). As a result, monitoring of the recovery process at the ecosystem level must be carried out in partnership between managers and research ecologists.

5. REFERENCE SITE SELECTION, UNCERTAINTY, AND ECOLOGICAL SIGNIFICANCE

5.1. Selection of Reference Sites

The selection of reference sites for recovery analysis must involve consideration of assessment endpoints. In cases where disturbance-related effects are minor, reference sites can be drawn from nearby sites, from data on conditions before the disturbance, if available, or from similar but spatially distant sites (e.g., chaparral sites in Italy as reference sites for recovery of disturbed sites in California). Data on the prior state of the study site is probably the best source of reference sites. Unfortunately, the availability of data depends on the extent and distribution of suitable monitoring programs. Similar sites in close proximity are probably the second best source of reference sites. Distant sites can be useful when the affected site is a rare ecosystem locally; however, comparisons must be based on collective properties rather than specifics (e.g., species lists) because taxononomic overlap is likely to be limited.

When disturbance is significant for the particular ecosystem, reference site selection is more complex. Under the best circumstances, knowledge of the natural recovery sequences for the ecosystem should be used to assess potential endpoints. Because successional trajectories are stochastic, however, recovery of disturbed sites also can be assessed statistically, if knowledge of state variables over successional time can be estimated (Loehle and Smith, 1990). This approach can be taken even if comparable control sites do not exist (Loehle et al., 1990).

Acceptability of recovery can be more heavily weighed if recovery is based on return to some point on a natural recovery trajectory rather than a return to a state identical to that present before the disturbance event. Determinations about acceptability, therefore, must include consideration of the frequency distribution of various states in the larger landscape. In some cases, a disturbance is capable of returning a system to an earlier or different stage without risk. Thus, remediation may be unnecessary if the stress mimics natural processes and the disturbance is incorporated in the larger landscape (Urban et al., 1987).

5.2. Sources of Uncertainty in Endpoint Selection

Several aspects of uncertainty relate to statistical uncertainty or the inability to predict recovery trajectories or endpoints precisely because of a lack of knowledge about biological functioning. Another source of uncertainty is based on ecological theory, which holds that because communities

and ecosystem are flexible they are not totally predictable. Several factors that contribute to uncertainty in recovery analysis are briefly described in this section.

5.2.1. *Species Substitution*

Species membership in an ecological community is determined by many factors, generally resulting in a wide variety over time and space. Thus, recovery criteria that specify exact duplication of community composition (in relation to a reference site) may be unrealistically stringent, of doubtful ecological necessity, and unachievable. An alternative, more realistic approach might be to consider community structure in a hierarchical manner based on ecological function. Recovery criteria might then specify that some subset of an ecological guild (e.g., insect pollinators) be reestablished, but not all species that are characteristic of the reference site. This approach assumes substantial knowledge about the ecosystem and the appropriate resolution of guild structure. In communities with many mutualisms and a complex guild structure, criteria for successful recovery may be particularly demanding. Conversely, in recently assembled communities (e.g., deciduous forests in the eastern United States [Davis, 1981]), guild structure may be more simple and recovery more easily achieved. Regardless, measures of species diversity, however sophisticated, will not adequately reflect the guild structure or recovery to the reference condition.

Given our current level of knowledge about the uniqueness of niches of individual species and uncertainty about the role of functional redundancy in communities and ecosystems, some ecologists would argue that anything short of exact restoration of the original species complex is unacceptable. The contention holds that some species—based on their ecological function—may be more important than others and that some ecosystems may require a more complete restoration of the original community in order to restore ecological functioning.

Introductions of exotic species seem to support the contention that some species are more important than others in particular ecosystems. In most cases, introductions have resulted in marked changes in community structure and ecosystem functioning. For instance, the introduction of exotic fish in Florida and the Southwest has resulted in dramatic ecological change (Courtney et al., 1974; Minckley, 1973; Moyle, 1976), despite the fact that introduced species may have comfortably fit into existing guilds.

Because emergent characteristics of introduced species cannot be predicted from biological information alone, recovery of communities should be assessed and managed using native species alone. Substitution of exotics for native species should be avoided as both a management tool or an

acceptable endpoint of successful recovery. While this principle has been widely accepted as valid for animal species, managers have been curiously slow to accept it for plants, despite extensive supporting evidence (e.g., studies concerning introductions of kudzu, melaleuca, brome, and prickly pear).

Managed ecosystems are often replete with human engineered species (e.g., in agriculture) or introduced exotics (e.g., pine plantations, reservoirs). In the Southwest, for example, although large lakes are rare, reservoirs are common and of recent origin, with fish communities that consist almost totally of exotics from the Mississippi River drainage. These recently assembled communities fluctuate widely. Moreover, the community and trophic structure is simplified and system-to-system variation is considerable.

Although establishing an assessment endpoint with reference to natural systems is difficult, substitutions that involve exotics should be avoided.

5.2.2. *Redundancy*

Redundancy concerns the concept that ecological function is often vested in several species. Based on this concept, reference systems should be selected that have a similar guild structure to the system at risk, suggesting that species membership in these guilds is a lesser concern. There are pitfalls to this approach, however. If the assessment endpoint is established with reference to ecological function (e.g., nutrient retention, erosion resistance), recovery may be achieved long before community structure is restored. If, on the other hand, the assessment endpoint is cast in terms of community composition, measurement endpoints must be developed along the same lines.

5.2.3. *Ecological Equivalence*

Ecological equivalence is a measure of redundancy that indicates the potential for species substitutions. The concept of ecological equivalence is based on the observation that similar ecological roles are played by quite different species in spatially distinct, nonoverlapping communities. This does not imply, however, that equivalent species can be interchanged (i.e., they may not be equivalents in a different ecosystem).

Along with supporting the argument against introductions, equivalence also dictates that reference systems must be selected from a broad range of possibilities, if assessment endpoints focus on ecosystem—rather than community—characteristics. For example, streams in the Sonoran desert of the southwestern United States may be suitable reference systems for streams in southern Spain, for

which there are no intact reference sites. Although species composition may show no overlap, recovery can be assessed in reference to guild structure and ecosystem function across these disparate sites.

5.2.4. *Successional Determinism*

Successional determinism refers to the predictability of successional trajectories and endpoints of apparently identical areas. For example, can a disturbed system be expected to regain its former structure and function? Is it likely to recover to a state similar to nearby but undisturbed sites?

The predictability and repeatability of endpoints are implicit in early studies of succession (Clements, 1916) and are now aspects of ecosystem development concepts (Odum, 1969). In terms of certain collective properties, the predictability and repeatability models that have been developed are adequate. For example, they show that biomass increases and P/R rises, then declines, during succession. A more realistic view, however, is that succession is flexible and stochastic, since trajectories of recovery are predictable only within a broad range and many alternative edaphic climaxes exist. Thus, rather than selecting a single site as a reference, the reference should account for a probability distribution of alternatives. Such a state-frequency expression should be derived from a knowledge of the recovery process at similar sites and study of the various seres typical in the particular region.

5.3. The Ecological Significance of Recovery

The ecological significance of recovery has three aspects, and establishment of assessment endpoints should treat each of these thoroughly. The first is the value of the affected system as a habitat, requiring that the risk assessment team consider recovery in terms of community structure. The level of detail at which this analysis should occur and the selection of endpoints will depend on whether system values call for high general diversity, the existence of endangered or threatened species, commercially valuable species, or keystone species. If general diversity is considered a value, for instance, then species diversity (i.e., richness and equitability components) should be monitored. When one or a few species are of special importance, individual populations must be monitored closely. Indicator species are likely to be of limited value in either case, since communities are notoriously flexible.

Recovery also is significant in terms of ecosystem services, which the recovery analysis must identify and then link with a suitable measurement endpoint. For example, wetlands are often valued

as nutrient filters because they can store material of interest (e.g., sediments or phosphorus) or shunt certain materials to the atmosphere (e.g., nitrogen, sulfur, carbon). Recovery of a wetland's retention capacity can be monitored as storage (e.g., sediment or phosphorus); upstream-downstream transport difference; or, assuming inputs are unchanged, output rate (e.g., flux or concentration). Relay filtration (the wetland's shunting capacity) cannot be satisfactorily deduced from input-output differences since these reflect storage as well. Thus, measurements of microbial processes such as denitrification, sulfate reduction, and methanogenesis are required. In some cases, simple measures such as redox potential may be correlated with these microbial processes and can be incorporated in the array of measurement endpoints adopted, but guidance from an experienced microbial ecologist is essential.

Finally, recovery analysis must include consideration of the ecosystem's habitat and services in the larger context of the landscape. To do this, the risk assessment team must know the extent to which the system of interest is linked with nearby or adjacent systems. For example, clear-cutting disturbs not only forests, but also streams and downstream systems such as lakes, reservoirs, and estuaries. No recovery is complete until linkage systems also return to reference conditions. Thus, measurement endpoints must either be extensive (i.e., total ecological monitoring of linked systems) or focus on the linkages themselves (i.e., transport rates linking systems), regardless of whether an ecosystem or a community approach to recovery is followed.

For all three aspects of significance, assessments must be made at several scales. At the landscape scale, for instance, disturbance is a desirable, natural process that contributes to the structural and functional diversity of the whole. Part of wise management involves ensuring that disturbance and recovery continue to occur at the appropriate spatial and temporal scales of the landscape. Thus, measurement endpoints at the larger scale must incorporate the concepts and tools of landscape ecology (Turner and Gardner, 1991).

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Issue Paper
on

UNCERTAINTY IN ECOLOGICAL RISK ASSESSMENT

Eric P. Smith
Department of Statistics
Virginia Polytechnic Institute and State University
Blacksburg, Virginia

H.H. Shugart
Department of Environmental Sciences
University of Virginia
Charlottesville, Virginia

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U.S. Environmental Protection Agency

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1. INTRODUCTION

Any scientific inquiry involves knowledge and lack of knowledge. The process of discovering what we know often leads to a better understanding of what we do not know. This general lack of knowledge or lack of certainty is what we refer to as uncertainty. In any ecological risk assessment, we try to obtain as much knowledge as possible about the relationship between exposure to the stressor and the response of the exposed system to the stressor. Because of lack of knowledge about how the particular system functions, how the stressor relates to the system, and how the system affects the stressor, uncertainty is generated. As described in the *Framework for Ecological Risk Assessment* (U.S. EPA, 1992) ". . . the uncertainty analysis identifies and, to the extent possible, quantifies the uncertainty in problem formulation, analysis and risk characterization."

Uncertainty in an ecological risk assessment refers to anything in the assessment that causes prediction to err, leading to doubts in the development or the results of the assessment. Lack of certainty in ecological risk often is associated with variability. Variability represents only one component of uncertainty, however. It is the difference between what is expected and what actually occurs. As it is typically used in statistics, variability is measured and interpreted assuming that the model and assumptions in the analysis are correct. Uncertainty is more significant since it also is associated with lack of knowledge; for example, in the choice of the model of the relationship between a stressor and an organism or lack of knowledge of parameter values in a model.

O'Neill and Gardner (1979) focus on three sources of uncertainty: model structure, parameterization, and stochasticity. Model structure uncertainty includes all unknowns in the process of developing a model. Choice of endpoints, determination of effects and relationships between the stressor and the ecosystem, and the selection of the model (if possible) are all included in this component of uncertainty.

Parameter uncertainty refers to uncertainty associated with the parameters of the model, given that one can be developed. Parameters must be estimated from laboratory, field, or other studies. Thus uncertainty is introduced by using an estimated value, which represents insufficient or unreliable information (data) for the parameter of interest. Further uncertainty may be introduced by recognizing that the value of the parameter may not be fixed but may vary (e.g., spatially or temporally).

Stochasticity means the natural variation in a system or response that is attributable to uncontrolled factors other than the stressor. Such factors are considered random in nature—examples include site-to-site differences, weather effects, changes in physical and chemical conditions, and

variation in the responses to the stressor (e.g., uptake of a toxicant)—and may or may not be included in the assessment.

Recognizing the three components of uncertainty is important because they provide an approach for characterizing uncertainty. Further, including stochasticity as a component implies a limitation to the risk assessment; that is, there is a point at which errors attributable to the other components become unimportant given the stochasticity. One may have excellent information about how a toxicant affects an aquatic organism, but the information will yield limited predictions because the natural variability in a stream has a strong effect on the exposure of and response to the toxicant. Stochasticity also allows for the occurrence of surprise events. Thus, despite a good knowledge base, the chance occurrence of certain factors may result in unpredictable events.

These three components of uncertainty in ecological risk are not, of course, the only views of components of uncertainty. A statistical view divides uncertainty into bias and variance. Bias is associated with ignorance about the model and parameters, while variance is usually associated with error resulting from natural randomness and model components that are not of interest. This is a useful categorization for investigating the predictive ability of a chosen model. An analysis of bias focuses on why a prediction deviates from expectation, and variation quantifies the uncertainty about predicted values. Suter (1990) prefers a larger categorization and considers uncertainty attributable to mistakes, such as errors in sampling, data entry, and computer coding. Surprises and mutations may form another component. One value in thinking about the first categorization, however, is that it provides a quantifiable view of uncertainty, and methods described in this paper are available for assessing the effects of these uncertainties on estimates of risk. Rather than concentrating on the various components of uncertainty, however, this paper addresses how uncertainty is involved in the risk assessment process.

Uncertainties are present at all phases of the risk analysis process. Table 8-1 lists some of the uncertainties in ecological risk assessment and their potential influence on assessment of risk. For example, the problem formulation stage of risk assessment has uncertainties that relate to the conceptual model and choice of endpoints. While in some problems it is quite clear what the endpoints are, what the stressor is, and what is at risk in the ecosystem, it is more common that some or all of the relationships are not known completely. For most risk assessments, uncertainty is associated with the selection of the model and even the modeling process. In systems with great knowledge gaps, the use of quantitative models may not be possible and simple qualitative models are the best that can be used.

Table 8-1. Uncertainties and Their Importance in Ecological Risk Assessment (adapted from Cothern, 1988)

Source of Uncertainty	Importance	Magnitude of Effect
Poor knowledge of system	Without any knowledge of the system, it is not possible to build a useful model	Many orders of magnitude
Extreme variation, incorrect scales	Great variation in weather, for example, may cause a large change in the importance of the stressor. Modeling large-scale phenomenon using a small-scale model may lead to great uncertainty	
Wrong model, endpoints, exposure routes	Measuring the wrong endpoint may lead to missed effects. Lack of knowledge of the exposure or model may lead to large errors	
Surprises	Unexpected effects may occur caused either by important gaps in knowledge or by random effects. Despite low probability of occurrence, effects can have great consequences	
Data collection practices	Errors in data collection and entry may lead to mistakes in interpreting statistical analyses	Order(s) of magnitude
Design of laboratory experiments and quality control	Adherence to laboratory standards is necessary to avoid errors induced by lack of care	
Variability in mesocosms or other ecosystem surrogates	Mesocosm studies have higher variability than laboratory studies and need to be carefully designed	
Extraneous variables	Physical conditions may have a strong effect on laboratory results	
Mistakes in statistical analysis	Outliers, wrong statistical model	

Table 8-1. (Continued)

Source of Uncertainty	Importance	Magnitude of Effect
Interactions	Uncertainty may be introduced by failing to account for interactions among species or combined effects of chemicals or other stressors	
Parameterization of computer model	Parameter estimates are taken from the literature, not from a fit to actual observations	
Mistakes in computer code of simulation models	Errors in code may lead to gross prediction errors	
Extrapolations across one species to another species to community or from laboratory to field spatial scale (local to regional)	Using a model developed for a simple endpoint may lead to errors when applied to estimate a more complex endpoint	
Spatial and temporal variability	It is difficult to predict with precision either over long time periods or over space	
Variability in laboratory test conditions	Variation in test organisms or concentrations of chemicals, for example, may cause under- or over-estimation of effects	Up to one order of magnitude
Minor mistakes in choice of statistical model	Including variables that are not necessary in the model may lead to increased variance; missing variables may add a bias	
Statistical design of manipulative studies (choice of stressor levels, randomization, number of experimental units, number of units per treatment)	Proper statistical design is important in laboratory and field studies especially when sample sizes are limiting. Estimates of quantities, such as No Observed Effect Levels, may be greatly affected by sample size and other factors	Potentially of great importance
Design of field study	Haphazard design of field studies may lead to incorrect decisions regarding effects	

Uncertainty is also present in the analysis phase of the assessment. Analysis of the exposure and the ecological effects are based on empirical evidence that may come from laboratory studies, simulation models, and field studies. The studies, however, may be from different locations, may involve different species or communities, and may simplify the actual relationships.

Uncertainties present in the analysis phase of risk assessment lead to vagueness in the characterization of risk. In this phase of the analysis, exposure and stressor response characterization are combined and interpreted in the context of the ecosystem. Thus, lack of certainty in analysis will lead to uncertainties in interpretation of risk. Additional doubt may be introduced in the interpretation of results by shifts in societal values. For example, the loss of certain species 20 years ago was not as important as it is today.

Why should the risk assessor or risk manager be interested in uncertainty and its analysis? For the following reasons:

- Because assessment of risk is a complex process that requires decision making, and uncertainty analysis can support the decision-making process. For example, because in assessing risk, information from different sources often is combined, knowledge about uncertainty associated with the sources can be important for weighting the information.
- Different approaches to modeling risk will involve different uncertainties. Because different experts may propose different models for evaluating risk, the risk assessor can use uncertainty analysis to identify the best approach or to determine whether approaches actually differ.
- Uncertainty analysis is useful for identifying a model's weaknesses and then improving the models.
- Uncertainty analysis can support decision making about remedial actions focused on the ecosystem. For example, different remediation scenarios might promise similar results but involve different uncertainties. Or, where different scenarios involve different costs, if uncertainties are similar, then the lower cost approach might be more appropriate.
- Quantitative estimation of uncertainty can help the risk assessor to decide whether more information needs to be collected, and, if so, which and how much information is needed.
- Uncertainty analysis is useful because it can lead to interval estimates of risk. Thus, rather than estimating risk as a single number, it may be described by a range or by using a distribution.

In our opinion, uncertainty should be assessed at all phases of the risk assessment, because knowledge and lack of knowledge are not separate components of the assessment. While studies often express confidence that the assessment approach taken is the best one, expressing any lack of confidence is also important so that a better assessment can be made given the state of knowledge. Because uncertainty is inherent in risk, understanding the points at which it comes into play and why will lead to risk assessments that are more accurate.

This paper examines the various phases of risk assessment in relation to uncertainty. Section 2 describes how uncertainty is involved in the process of problem formulation, addressing problems associated with model structure, scale, ecosystem characterization, and stressor characterization. In section 3, aspects of uncertainty in the analysis phase of assessment are discussed, particularly concerns arising from the analysis of statistical experiments and field studies, from extrapolation over different ecological and environmental scales, and from the implementation and parameterization of models. Section 4 describes how knowledge deficiencies enter into the characterization of risk and how effects of uncertainty are evaluated quantitatively and qualitatively. Several methods for dealing with doubts in model structure, statistical models, extrapolations, and computer models are presented. The integration of uncertainty into the decision-making process, another important consideration, is described in section 5, along with some suggested approaches. Our purpose is not to present an exhaustive review of all sources of uncertainty and methods for its assessment. Rather, we have attempted to present some of the important aspects of uncertainty as they relate to ecological risk assessment and the Framework, while offering suggestions for dealing with lack of knowledge.

2. PROBLEM FORMULATION

Problem formulation can constitute a major source of uncertainty in the fundamental predictions used as a basis to assess risk (table 8-1). This uncertainty can arise in the mathematical structures used in developing predictive models and from the implementation of models once they have been developed. Model development is often a surprisingly ad hoc procedure, and the consequences of selecting one formulation over another are not always given a particularly thorough analysis. Frequently, anticipated parameter estimation problems influence decisions in model formulation.

For example, a typical model for demographic analyses would be in the form of ordinary differential equations for each population's numbers. Such models are formulated for demographic problems because there is a tradition for estimating their parameters (natality, mortality) and because the more complex age- or size-structured integro-differential equations are difficult to parameterize and solve. If the population response of interest is dependent on the age-specific changes in the mortality rate, then the simple model will not have a structure that allows a realistic assessment of the consequences. One could develop equivalent examples involving tradeoffs in model development between model simplicity and model complexity or between spatial heterogeneity or homogeneity. Biology and ecology involve the relationship between geometrical structures of living things and attendant processes.

The problem formulation phase in ecological risk assessment almost always involves the identification of the parts of the ecosystem involved and the function of these parts. This fundamental theme in problem formulation in the biological sciences recurs in cellular biology, genetics, morphology, and population structures as the relationship between form and function. The organization of tissues is intimately related to their physiological function. Moreover, it is important to the function of the DNA molecule that it is a spiral helix. The adaptive implications of the morphology of plants and animals is central to both taxonomy and evolutionary biology.

In biology, these themes are variously described as "structure and function" or "pattern and process." Depending upon the examples chosen, there may be an emphasis on the manner in which processes influence pattern. For example, what changes in the morphology of the vertebrate limb develop in the evolution of flight? Or, how do the seeds of plants vary between arid and moist environments? In other examples, the emphasis is on the pattern or structure modifying the processes. Pattern and process are linked in a biological yin and yang in which each causes and is caused by the other.

The relationship between form and function, or pattern and process, is also a classic ecological theme (Lindeman, 1942; Watt, 1947; Whittaker and Levin, 1977). Bormann and Likens (1979) pointed out the effects of changes in forest structure on processes such as productivity and nutrient cycling. Many ecologists recognize that pattern and process are mutually causal, with changes in ecosystem processes causing changes in pattern and modifications in ecosystem pattern changing processes. Nonetheless, it is difficult to investigate directly the feedback between pattern and process.

In many ecological studies, a knowledge of the degree of dominance of particular causal factors at particular scales is especially important. The knowledge of which factors are important at a given scale also is involved in the determination of the "rules" for deciding what should be included in the formulation of a given model. That different phenomena may be invoked in developing models of analogous phenomena at different scales is responsible to a degree for what is categorized as the "art"—as opposed to the science—of ecological modeling. While a determination of the significance of processes at a given time or space scale is important in model formulation and evaluation, it is a consideration neither trivial nor unique to the developers of computer models. For example, in the case of "hierarchy theory" (Allen and Starr, 1982; Allen and Hoekstra, 1984; O'Neill et al., 1986; Urban et al., 1987), one finds a focus on expressing relevant mathematical developments in a manner that can provide insight into the ways ecosystems are structured at different scales.

No hard and fast rules for the "proper" method of model formulation exist. At the fundamental level, science is a creative endeavor, and model formulation is not immune to this need for creativity. Nevertheless, certain considerations and procedures have proven efficacious in developing models and for identifying uncertainties, as discussed below.

2.1. Structural Uncertainty

Discussions of uncertainty in models often are restricted to problems of parameter estimation from observations of a given quality. Important sources of uncertainty arise in the formulation of a particular problem. Currently a dichotomy between process and structure is an important limitation to a comprehensive synthesis of the behavior of ecosystems (O'Neill et al., 1986). Are ecosystems an interdigitation of processes accounting for the dynamics of water, carbon, and mineral elements, or are they a structured collection of organisms interacting with their environment? While this distinction is never quite this clear in ecosystem research, it tends to be so in an important class of synthesis tools—simulation models. Current ecosystem simulation models can be categorized as either process or structure models. Process models attribute cause in ecosystem responses almost entirely to

processes, while little explicit importance is attributed to system structure (Parton et al., 1988). Models of ecosystem structure tend to represent the opposite conceptualization (Shugart, 1984).

Terrestrial ecosystem structure, provided primarily by the vegetation, is the result of responses of individual plants to environmental constraints of resource availability, temperature, and disturbance. Ecosystem structure provides a context for other processes and, in turn, is modified by them. The response times of structural and process dynamics can be quite different, with processes typically having faster dynamics than structure. This temporal mismatch makes it difficult to reconcile the structural and process aspects of ecosystems and it creates a dichotomy in the thinking of ecologists that is a consequence of differences in embedded scales of the phenomena they consider.

2.2. Ecosystem Characterization

In a classic paper by Tansley in 1935, the term "ecosystem" was first defined as an arbitrary system with respect to both its spatial extent and the phenomena considered:

> The more fundamental conception is, as it seems to me, the whole system (in the sense of physics), including not only the organism-complex, but also the whole complex of physical factors forming what we call the environment—the habitat factors in the widest sense. Though the organisms may claim our primary interest, when we are trying to think fundamentally we cannot separate them from their special environment, with which they form one physical system.
>
> It is the systems so formed which, from the point of view of the ecologist, are the basic units of nature on the earth. These ecosystems, as we may call them, are of the most various kinds and sizes. They form one category of the multitudinous physical systems of the universe, which range from the universe as a whole down to the atom.

In this first use of "ecosystem" in the English language, Tansley stressed that ecosystems are of "various kinds and sizes." This relative arbitrariness and abstraction was viewed by Tansley as a necessary step in the formulation of an ecological science on a par with physics and other, more established sciences. The usefulness of the ecosystem concept has been proven in the 50 years since Tansley coined the term.

As more fully defined by Tansley, an ecosystem can be depicted as the set of interacting external variables, ecological processes, and patterns all with equivalent space-time domains. An ecosystem may be large or small with respect to either temporal or spatial scales, but there should be an equivalency in the time and space domains in the patterns and processes considered.

2.3. Stressor Characterization

The categorization of controlling factors important at different space and time scales in particular ecosystems has been the topic of several reviews (Delcourt et al., 1983; Pickett and White, 1985). Historically, this interest is evident in A.S. Watt's early work (1925) on beech forests, which he elaborated on in his now classic paper on pattern and process in plant communities (Watt, 1947). These themes have been reiterated by several subsequent ecologists (Whittaker, 1953; Bormann and Likens, 1979).

The factors governing structure and processes vary considerably within and among biomes. In mesic forests, a frequent constraint is the availability of light. As a forest environment tends from mesic to xeric, or nutrient-poor conditions, the effective constraint shifts from above- to below-ground factors (Webb et al., 1978; Tilman, 1988; Smith and Huston, 1989). Under still drier conditions, forest changes to grassland in which the principal constraint is below ground, suggesting patterns in the influence of environmental constraints in structuring ecosystems across broad environmental gradients.

The importance and nature of ecological disturbance has been a determinant of ecology research for the past 20 years. Disturbance is an ambiguous term in the vocabulary of ecologists (Sousa, 1984; Pickett and White, 1985); in practice it is necessary to define the spatial distribution, frequency, predictability, size or area, intensity or severity, and interactions with other disturbances to sufficiently specify a particular disturbance regime (Pickett and White, 1985). A disturbance event is usually thought of as a discontinuity in an exogenous variable(s) that can alter system structure and the balance among process rates, potentially leading to system reorganization. This definition is directly analogous to the stressor terminology used in the Framework. Disturbance has been variously viewed as an external catastrophe, a rejuvenating force, a source of diversification, and an active principle counterbalancing the tendency for competitive interactions to lead to the eventual extinction of species. In these roles, disturbance is characterized as external (i.e., as a force from without that alters the ecological system in some way) or, in some cases, internal (i.e., a force that is at least in part mediated in its frequency or effect by factors inside a particular ecosystem). Of course, disturbance is all of these depending on point of view and the disturbance in question.

Temporal patterns in disturbances may have profound effects on ecosystem structure as well as process rates. For vegetation structure, the interplay between disturbance features and plant life-history traits is the important feature (Denslow, 1980; Sousa, 1984), while for system processes, the critical features are those that are rate mediating (Odum, 1969; Vitousek and Reiners, 1975;

Bormann and Likens, 1979; Vitousek et al., 1989). Disturbance frequency may resonate with natural frequencies of plant growth forms (i.e., phenology, time to first reproduction, longevity) to amplify environmental patterns (Neilson, 1986). Thus, there may be an interaction between abiotic environmental constraints and biotic natural-history phenomena that results in system-specific spatial and dynamical patterns (Allen and Wyleto, 1984; O'Neill et al., 1986; Urban et al., 1987; Shugart and Urban, 1989).

2.4. Conceptual Model Formulation

The interweaving of models of differing fundamental scales is a problem of considerable difficulty. In many fields using dynamic models, so-called stiff system problems, in which the time constants for important processes span several orders of magnitude, can exceed the capacity of modern digital computers. Interestingly, both production and decomposition modeling can lead to stiff systems of equations if fast processes that influence production and decomposition (e.g., biophysical responses of the leaf surfaces with responses of sec-1 or min-1; microbial growth or soil chemical kinetics with similarly fast responsivities) are coupled with slow processes (e.g., tree mortality, soil organic matter dynamics, or soil genesis). As a numerical problem, stiff systems are sometimes solved by separating the system into one system of "fast" variables and another of "slow" variables that can be evaluated separately. In ecological modeling, this same procedure is applied in something of an ad hoc manner in the assumptions regarding which processes can be included or excluded from a given model formulation.

One would hope that the relatively fast-response models can be interfaced to some degree with relatively slow-response models. For example, figure 8-1 is a diagram developed at a National Aeronautics and Space Administration (NASA) conference on the topic of scale considerations in the interfacing of climate models and ecological models. It was developed in part in response to climate modelers interested in having dynamic models on the response of vegetative canopies at a space scale (about 100 x 100 km^2) that exceeds that of physiologically based models (and, indeed, is on the outer fringe of the spatial domains of even the larger scale ecological models) but is on the time scale of many physiological models (approximate minutes and hours). While it is clearly important that ecologists in developing vegetation models have an initial interest in understanding the time and space scales of ecological phenomena, we must also realize that scientists in other fields (e.g., atmospheric sciences, oceanography) are increasingly posing modeling problems for ecologists that are in unfamiliar parts of the space-and-time domain.

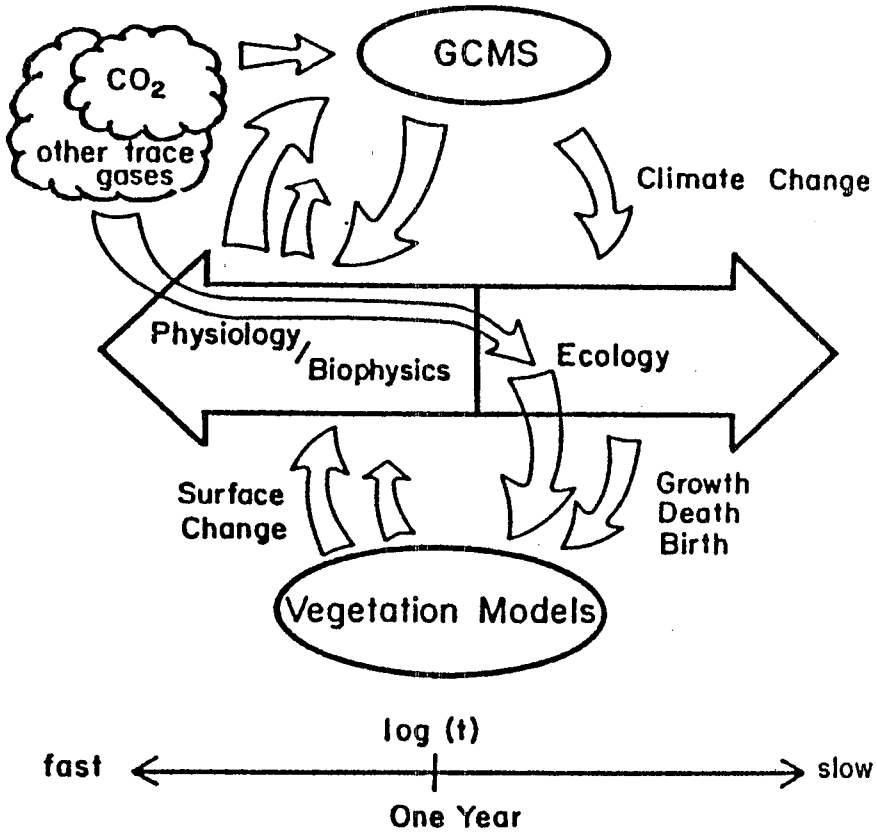

Figure 8-1. Links among various atmosphere-biosphere modeling efforts (Shugart, 1986).

In many cases, one would like to use a model in "scaling-up"—communicating in some quantitative and relatively direct fashion the outcome of a model at a relatively fine space or time scale to a coarser scale. In many physically based systems, scaling-up can be done in a relatively straightforward and, in some cases, elegant fashion because one can assume (or demonstrate analytically) that certain terms of the equations can be ignored at larger scales of time or space. Campbell's text (1977) on environmental biophysics provides a discussion of the relation among the Penman-Monteith equation for evapotranspiration and several radiant-energy-driven, coarser-scale evapotranspiration models (e.g., the Priestly-Taylor equation [1972]) that serves as an excellent example of this sort of scaling-up. Unfortunately, the problem of scaling-up from detailed models can be considerable in some cases. Three important classes of problems include:

Numerical problems in scaling-up. In the case of the scaling-up of a detailed model, such as the type of models that were classified as "physiologically based" models, one is faced with problems such as propagation of error, nonconstancy of conditions assumed to be constant over relatively short (or small) measurement intervals, or the computational cost of solving equations with small time constants over long intervals. This last problem can, in extreme cases, lead to numerical solution problems as well. Of course, physiological models are constructed at space-and-time scales appropriate to benchtop experimentation and thus are potentially valuable for interpreting experimental data on novel conditions such as alterations in the carbon dioxide concentrations.

Initial condition sensitivity and chaotic behavior. One important development in mathematics that has implications in the scaling-up and development of detailed models has come from the analysis deterministic systems that are chaotic or unpredictable. For example, starting with two sets of conditions describing the state of the atmosphere that are so similar as to be identical if one attempted to measure them, the equations describing the fluid motion of the atmosphere have the property that the predicted atmospheric dynamics will diverge over time. This has made the long-term prediction of weather using detailed physical models appear to be impossible except in a statistical sense. Thus it is reasonable to expect such dynamics to prevail in natural ecosystems (Whittaker and Levin, 1977).

Transmutation across scales. O'Neill et al. (1986) used the term "transmutation across scale" to describe the tendency for representations of processes to be transmuted or changed when viewed from a different point of reference. As an example, figure 8-2 shows the proportion of trees in different

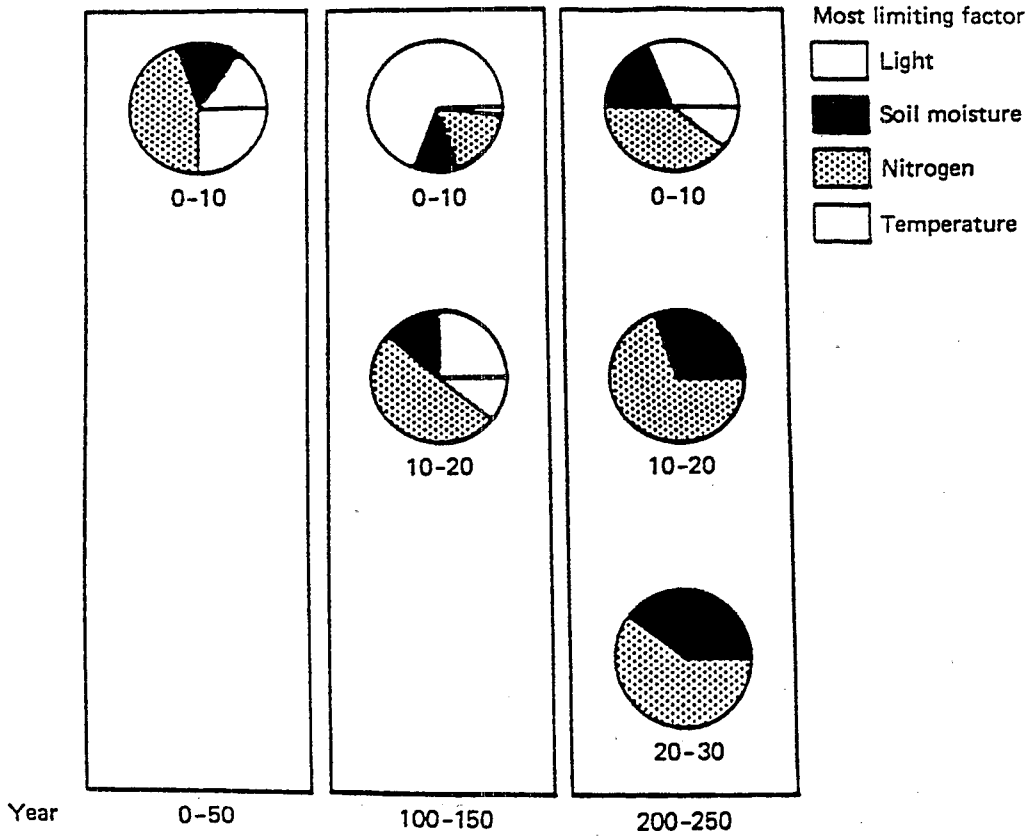

Figure 8-2. **Most limiting factor in a simulated successional sequence as a function of tree height and time. The simulated successional sequence is based on results from an individual tree-based model of a northern hardwood forest. The simulation is initiated at year 0. The sections of each circle are allocated according to the proportion of individual trees having the indicated factor as the most limiting factor to tree growth (adapted from Pastor and Post, 1986).**

strata and in stands of different ages limited in their growth by light, water, temperature, and nutrients. The result was obtained by running an individual tree-based simulation model from an open (bare ground) condition to a relatively mature forest. The interesting feature of this result is with respect to the so-called Liebig's Law of the Minimum when applied across scales. Liebig's law states that the most limiting factor will control growth. Pastor and Post (1986) used the LINKAGES model to evaluate the principal constraints in forests through time and among trees of different stature. The principal constraint changed from below ground to above ground as trees grew and the canopy closed. The principal constraint also varied with tree size and time in the simulated succession. The dominant individuals were limited by below-ground constraints, while understory individuals shifted from below- to above-ground constraints as they were overtopped. This result cautions against oversimplifying ecosystems in terms of "the primary constraint" and suggests parallels in patterns of constraint through time and over spatial gradients.

The point here is not to defend Liebig's law or to speculate at what level (if any) it should be applied. Rather, the point is that the "law" does not work at the level of the individual and the canopy of a forest simultaneously. Indeed, any number of rules for allocation or optimization have this property. An optimal solution for the shape, function, or cost of a part of a system rarely can be expected to conform to the shape, function, or cost of the part when the entire system is optimized by the same criteria.

2.5. Implementation of Models

Several studies have attended the natural scales of ecosystems in a qualitative or semiquantitative manner (Delcourt et al., 1983; Urban et al., 1987). By "natural scales," we refer to the temporal frequencies or patterns of spatial variability to which a particular system is responsive. Obversely, this implies that the smaller scales to which the ecosystem cannot respond (i.e., higher frequencies and finer grained spatial patterns) are witnessed only as average values and give an indication of the larger scales that the ecosystem witnesses as effectively constant values. The structure of a given model implementation reflects fundamental decisions as to what state variables and parameters should be included in a given model.

The natural frequency of a system can be determined through spectral analysis (Shugart, 1978). Emanuel et al. (1978) and Shugart et al. (1981) computed power spectra for the FORET model and found a dominant peak at a periodicity of about 250 years, which roughly corresponds to the lifespan of trees and reflects the pulsing of forest systems by the death of large trees (i.e., gap dynamics).

The use of these sorts of analytical techniques and the insights that they provide into fundamental system dynamics have been proposed as a theoretical alternative to ecological modeling (Pielou, 1981). Little direct work has been performed on the spectral performance of actual ecosystems, however—something that is largely attributable to the long, regularly sampled data sets that are needed to perform the analyses. Some model analyses (Emanuel et al., 1978) and analyses of data sets from microcosms (van Voris et al., 1980) have indicated a relationship between the spectral content and the time response of a system and the response of the system to stress. If this is the case, one might then have an a priori prediction of the relative stress resistance of ecosystems.

An additional consideration concerns the spatial expression of system dynamics. One technique recently presented by Levin (Levin and Buttel, 1986) is based on a regression of sample variance versus scale (as aggregate sample size). In this analysis, a temporal variance term is computed for some system attribute and recomputed for a geometric series of sample quadrants from a grid or transect. If both the variance term and sample size are log transformed, a linear relationship obtains, and if there is no spatial autocorrelation in the system (no scale-dependent interactions), the slope of this line is -1.0. Spatial autocorrelation (scale-dependency) deflects the regression slope from -1.0. The important feature of this analysis is that there is an expected "null model" for cases in which there are no spatial effects.

2.6. Summarization

The conceptual phase of ecological risk assessment is associated with summarizing information and making choices. In the summarization process, we discover how much we know—and often how little we know—about the stressor and ecosystem response. Uncertainties occur resulting from choice of endpoint, choice of model and modeling approach, choice of scale, and availability of information. Choice of endpoint, both for the stressor and ecosystem, are frequently governed by available information. Because ecosystems are dynamic in time and space, the issue of scale is especially relevant to the understanding of uncertainty.

A fundamental part of ecological risk assessment is the documentation of knowledge and justification of the model. Part of this documentation requires a discussion of the uncertainties related to endpoint, scale, process, and structure as well as other factors important to the modeling process. Ranking model components in terms of uncertainties, model and information flow charts, and description of information needs is valuable at this stage for making management decisions about research necessary to reduce uncertainty (see section 4.3).

3. ANALYSIS-PHASE UNCERTAINTY

In the analysis phase of the risk assessment, uncertainty arises from the process of transforming a general model(s) chosen in the model development phase into a specific model(s) and attempting to implement the model. Recognizing that different risk assessments may use different types of models and possibly multiple models, we describe the three main classes of models that we expect to be used in a risk analysis and address the associated uncertainties. These models are derived from designed experiments, observational studies, and biological principles. The first two types of models tend to be simple and empirical, while models derived from biological principles tend to be more complicated. Additional information on models is provided in other chapters (see chapter 4, on exposure assessment; chapter 5, on effects characterization; and chapter 9, on risk integration methods).

In ecological risk assessment, there are two main components to the analysis of risk: the exposure and the modeling of exposure, and the modeling of ecosystem effects given an exposure or exposure profile. Expected results from this phase of the assessment are profiles of exposure and effects. The role of uncertainty analysis in this phase involves the description of doubts in the assessment of their effects. We begin by discussing the two components of the analysis phase (see table 8-1), focusing on uncertainty. Then we describe unknowns associated with designed experiments and extrapolations, field studies and extrapolations, and computer models. The importance of evaluating model credibility is discussed in the last part of this section. Quantification of uncertainty in the profiles is described in section 4.

The analysis of exposure is an area where there may be a great deal of knowledge uncertainty. With pesticides or chemicals, uncertainty may arise because of interactions with other chemicals, variation attributable to peak versus averages, and spatial variability. Differences may be attributable to the frequency of exposure, duration of exposure, and intensity of exposure as well as the path of exposure. Interactions with nonstressors or with other stressors may enhance or inhibit effects. For example, surfactants bind to soil, thus reducing the exposure concentration in the water column. Increased total dissolved solids could produce a direct threat to aquatic biota, but also reduce concentrations of metals and organic contaminants. Some adjustment needs to be made for these secondary effects, since the measured concentration of material that enters a system may not reflect the exposure.

A significant source of uncertainty may be attributable to the inability to measure the stressor or exposure to the stressor. For example, chemical levels predicted to affect organisms may be below

detection levels. Also, lack of knowledge about when the stressor is important may lead to poor exposure information. For example, an important challenge to researchers sampling sites contaminated with toxic wastes is the problem of spatial variability. Moreover, knowing the location of "hot spots" is critical to obtaining information about the magnitude of potential exposure.

Models that predict exposure may range from simple regression models to complex computer models. With computer models, additional uncertainties may arise because of the difficulty in using available data to estimate parameters in the model. This is because unlike a statistical or empirical model, the parameters often are not directly estimated from observations; rather, estimates are frequently taken from other studies, possibly on other organisms, from other sites, or under other conditions, for example. Parameter estimates may be "fine tuned" to produce models that reasonably mimic actual data on expectations, with the "tuning" done by changing the parameter estimates until the model is close to observed data. This process is not necessarily unsatisfactory since the model (and its associated set of parameters) may be used as a "testable hypothesis" when validation data are available.

A similar set of uncertainties exists that is associated with models of ecological effects. Often there is a lack of knowledge about which organisms are affected and how the organisms are affected by the stressor. While primary effects on organisms often are reasonably predicted, subtle effects are rarely predicted with reliable accuracy. For example, available knowledge may indicate which species are likely to be harmed by a stressor, but the analyst may not have information on the effects of reduced abundance on species not harmed by the stressor (e.g., competition and predation effects).

Models of ecological effects range from simple to complex (see chapter 5, on effects characterization). Although simple models often are easy to develop and parameterize, extrapolations may be required to relate a model to an ecological problem. Difficulties also may arise that relate to variables not included in the model. More complex models may incorporate greater ecological realism at the expense of predictive ability. Difficulties in developing more realistic models include knowledge gaps associated with components of complex models (or an entire model), lack of data for parameterization of the model or components of the model, problems associated with incorrect scales, and problems associated with choice of model endpoints.

Even adequate knowledge of ecological effects and exposure does not necessarily guarantee a successful model. Complete knowledge is rarely available in ecological risk assessment, and any lack of knowledge in the exposure profile and the response profile may lead to surprises. In general, surprises are viewed as events that occur but are not predicted by the model, or events that have a

low probability of occurrence. Sometimes these events are not even considered in the formulation of the models. For example, one of the most thoroughly investigated ecological problems is the outbreak of the spruce budworm in Canada (Walters and Holling, 1990). Despite being well studied and modeled, the models and risk assessors failed to predict outbreaks in Newfoundland in the 1960s and increased frequencies of insects in stands of young trees.

Another example of surprise concerns ecological nonlinearities and capacities of soils. Stigliani (1988) discusses three surprises associated with shrinkage of capacities of soils: the release of toxics following cessation of liming; the release of toxics attributable to changing redox conditions in overlying water bodies; and the release of sulfates in wetlands under dry conditions.

It seems that little can be done to predict surprises. Recognizing the possibility of such events suggests that ecological risk assessors must adapt to new knowledge, make attempts to plan for surprises by recognizing knowledge uncertainties, and be willing to alter strategies as surprises occur.

3.1. Uncertainties in Designed Experiments

Models often are derived from the results of experimental studies that include bioassay studies, field experiments, and microcosm and mesocosm studies. These studies often investigate a relationship between a stressor and the ecosystem or a surrogate for the ecosystem. Because these are designed experiments, uncertainties arise from the statistical design, implementation of the design, and analysis of the resulting data.

Uncertainties may be introduced by choice of statistical procedure and design. Designs of experiments involve choice of hypotheses to be evaluated, choice of responses to measure, choice of stressor and magnitude of stress, and methods of analysis. Additional components include laboratory standards, measurement process, and quality control associated with the experiment. Improper control of the quality of the study may even lead to erroneous results. An important aspect of experimental studies is the control of uncertainty through the control of variation, which involves proper choice of sample size, proper choice of levels, and control of variation attributable to extraneous factors that may influence results. Careful attention must be paid to the design of the experiment to ensure the validity of the results.

Some uncertainties may arise from the interpretation of statistical data. A particularly important concern is the proper interpretation of statistical methods, such as hypothesis tests (Parkhurst, 1990), which are used to evaluate a statement associated with a stressor. For example, the system has recovered from stress, a dose of 0.5 mg/L has no effect on organisms, the stressor has no effect. In

scientific studies, hypotheses are usually stated to be rejected. The hypothesis of no change or effect is called the null hypothesis (H_0), while the hypothesis of effect or change is called the alternate hypothesis (H_1). The decision process is given below.

	Truth	
Decision	H_0	H_1
Do not reject H_0	No error	Type II error
Reject H_0	Type I error	No error

Errors can be made in rejecting or not rejecting the null hypothesis. For example, if the null hypothesis is that a particular stressor has no effect, the null hypothesis is not rejected if the variation in data is large relative to the signal in the data. The hypothesis is not assumed to be true simply because it has not been rejected. The study may not have been designed well enough to detect the signal that would show the hypothesis to be incorrect. The usual approach for assisting in the design of experiments involves the power of the test, which is the probability that the hypothesis is rejected when in fact the hypothesis is false. The proper statistical design of a study will focus on the power of the test and choose a sample size to adequately detect important signals. Peterman (1990) emphasized this point in evaluating the design of impact assessment programs for power plants. In many studies, Peterman found the sample sizes used were insufficient to assess anything but a gross change in ecological conditions. Hence, some of the studies may have indicated no effect attributable to the power plant when, in fact, one occurs. Even with adequate test power, detection of change or differences may be hindered by improper sampling, natural and unnatural influences, and confounding factors. For example, an effluent that is discharged into a rocky substrate may have no effect on the biota present because there are few biota present to affect. Factors such as floods may alter habitats in control and impact sites differentially, making them no longer comparable.

Hypotheses about recovery (see chapter 7), no effects, and the safety of stressors are difficult to evaluate from a purely hypothesis-testing approach. In such studies, it is desirable that the null hypothesis not be rejected. The above truth table may be represented in terms of ecological effect:

	Truth	
Decision	No ecological effect	Ecological effect
No ecological effect	No error	Type II error
Ecological effect	Type I error	No error

The difficulty arises because the rejection of a hypothesis is a strong statement, while evidence that favors the null hypothesis is regarded as confirmatory evidence and not proof. We do not, for example, prove safety in a statistical study. Thus the decision of no ecological effect, or "the system has recovered," really means that there is no evidence to indicate otherwise.

This consideration is important, for example, when estimating safe levels of toxicants or other stressors. One approach is to set up an experiment in which groups of organisms are exposed to different concentrations of a toxicant. Tests would then be conducted comparing the control (zero dose) to concentrations. If the effect of the concentrations was not statistically different from the control, then the concentration could be said to be safe. Only when the null hypothesis of no difference between the control and the toxicant concentration is rejected statistically could the concentration be viewed as unsafe. As pointed out by Parkhurst (1990), the burden of proof is placed on the toxicologist to show that a concentration is more toxic than the control, rather than being placed on those who would use the toxicant to show that a dose does not cause an effect. Human risk assessment concerning lead suggests that as more endpoints are measured, and as the ability to measure effects improves, estimates of safe levels become lower. Thus estimates of safe levels from simple studies must be viewed with some suspicion.

It is important, therefore, that gross uncertainties about hypothesis tests be controlled by proper statistical control of studies. Small sample sizes, poorly designed experiments, ignorance of the proper variables to measure or proper times to sample may lead to acceptance of a no effect assessment when in fact there is an effect and high uncertainty.

Although the above discussion focuses on hypothesis testing, many of the same comments and concerns pertain to the creation of models from experimental studies and the estimation of exposures and effects. Many of the difficulties with dose-response models, which are perhaps the most commonly used model in risk assessment, are discussed in the health literature (Cothern, 1988; Park and Snee, 1983). Choice of statistical model involves choosing a deterministic component to explain

relationship and a stochastic component to represent what is unexplained. Different modelers will make different choices that may be indistinguishable over easily measured regions but quite different in other regions. Examples include threshold models and models for low-dose extrapolation. Another example concerns choice of a test statistic to measure change, which in a hypothesis test implies choice of a measure of biological change. This is particularly a concern with multivariate data (multispecies data), for which it is common to use nonstatistical measures of distance.

Although statistical models produce estimates of uncertainty in the form of variance of predicted values and parameters, these estimates also have uncertainties. For example, different laboratories may produce different models and estimates. Effects associated with laboratory methods, technician effects, experimental conditions, and organism variation (table 1) may lead to significant doubts about the validity of these estimates. Further concerns arise in trying to extrapolate the results of a designed experiment made at one level (e.g., laboratory scale) to predict results at another level (e.g., field scale).

3.2. Uncertainties Attributable to Extrapolations

It is difficult (and in some cases impossible) to measure directly the effects of a stressor on an ecosystem. Just as we cannot allow a nuclear power plant to discharge radioactive material to see what happens, we cannot wait 20 years to observe the effect of low levels of pesticides on duck populations. Thus, given the need to assess effects and to model the response of stressors, surrogate studies and models must be used. For example, a model of the toxicity of a chemical may be developed based on the chemical structure and activity of the chemical. Similarly, the toxicity of a chemical to one species may be used to estimate the effect the chemical has on the growth of the species or on the growth of other species. When models are used in this manner, uncertainty arises that is attributable to the need to extrapolate across endpoints, species, and even systems. For example:

- Results may be based on small-scale studies and applied over a larger scale. (For instance, since a toxicity test often spans a short time period [e.g., 7 days], effects of low dose may be misleading because a long-term low dose may give a different effect than a short-term one.)
- The endpoint of interest may be replaced by a measurable endpoint. (For instance, laboratory studies often involve a mortality or growth in a single species, while the endpoint of interest may be a measure of ecosystem function.)

- The genetic composition of laboratory species may be different than in field species.
- Laboratory conditions may not reflect field conditions or may relate to factors not considered in laboratory studies (e.g., hardness of the water).

For additional discussion of this problem, see Suter (1993). An approach for estimating the effects of extrapolation is discussed in section 4.

3.3. Field Studies

Field studies often are more complicated than laboratory studies and include additional sources of uncertainty. One of the important components of the laboratory study is the ability to control variation and bias through randomization and replication: Randomization of levels of the stressor to the units being studied can eliminate potential sources of bias; replication of experimental units is a useful tool for reducing uncertainty attributable to variation.

Typically field studies are not controlled experiments. More commonly they are observational studies or at best partially controlled experiments. The stressor is not applied at random to the field; rather, more often it exists in the field. For example, in the case of the Northern spotted owl, where the stressor is the cutting of old-growth forest, sections of old- growth forest containing spotted owls are not cut at random. Other examples of what are actually observational studies include environmental accidents such as oil spills.

The difficulty with observational studies is that measured effects represent possible effects from the stressor as well as effects from any other stressors or agents that affect the system. While the different effects may be confounded in an observational study, confounding is less important in controlled experiments because of the randomization. In assessing the effects of cutting old-growth forest on spotted owls, it becomes difficult to assign a quantitative effect since tree cutting is confounded with other factors such as local climate changes, other changes in the forest that are specific to the cutting area or home range of the owl, and other potential stressors (e.g., other human interventions), as well as unseen influences.

Uncertainty in the results of field studies can be especially important when replication of the stressor is not possible. The most common of such situations is when the stressor is unique to the site. Consider, for example, assessment of the effect of a power or chemical plant on a river ecosystem. One approach to assessing an effect is to collect samples before and after the plant starts operation, above and below the site of the plant on the river. Suppose that a study finds that there are some minor differences in ecosystem organisms and that these differences change after the plant is

started. Several problems arise in trying to assign the cause of change to the plant, because the change also may be attributable to alterations in the habitat of the site from natural events (e.g., flooding) or to construction of the plant (e.g., from soil deposition caused by erosion), and not its operation. The change also might be associated with factors such as increased fishing in the vicinity of the plant or differences in the sampling protocol (Smith et al., 1993). Additional uncertainty associated with field studies is attributable to their observational nature (Hurlbert, 1984). Thus the results of a field study have to be viewed as associations between a stressor and the system and not linked to a cause. Complementing field studies with laboratory studies is an effective way to reduce some of this uncertainty.

Another concern associated with field studies, especially monitoring studies, is the lack of statistical rigor in the design and collection of the data (Rose and Smith, 1992). Often data are collected without an explicit statement of purpose (hypothesis) and with little regard for principles of statistical or sampling design. Mismatches between the monitoring design and hypotheses of interest result in loss of efficiency and power. Hypotheses may be poorly stated or may change over time, and data collected for one purpose may be used for other purposes (i.e., to address hypotheses they were not intended to address). Along with loss of test power, this mismatch between hypotheses results in increased uncertainty and potentially misleading conclusions.

Besides problems in interpreting stressor effects in field studies, problems also can be associated with establishing the exposure profile in the field. An important problem in risk assessment involves establishing the existence and a means of measuring the exposure, since exposure in an ecosystem is often patchy, making the integration of effects difficult. For example, because toxicants in an aquatic system may be diluted, the concentration depends on flow, location, and the time interval after the toxicant was introduced. Approaches have been developed for dealing with these difficulties. Models of exposure and effects can incorporate some of these effects. When information is available, models can be designed to focus on average exposure and effects. Also, field studies can be designed to detect exposures greater than a given magnitude with some certainty when averaging over time or space. A more difficult problem is designing studies that account for multiple effects and exposures.

3.4. Computer Simulation Models

Because of the complexity of ecosystems, computer models are often used to characterize the effects of stressors or to predict exposure profiles. Computer models have an advantage over statistical models in their ability to accommodate greater complexity; also, they are designed using

biological principles rather than empirical relationships. The additional complexity, however, may result in different—and possibly greater—uncertainties that arise from the need to parameterize and calibrate the models and to deal with stochasticity in the system.

Computer models often consist of components, each one of which deals with a different part of the system. For example, a model of watershed acidification may involve as many as 30 processes or subprocesses over one or more spatial scales. Each component is a small model that requires parameterization. When using these models to assess effects in an untested region, the parameterization may involve different soil layers and the assumption of steady-state conditions. Although such parameterization can be quite complex—involving hundreds of parameters—rarely is enough information available for estimating each parameter individually. Frequently in such situations, models are calibrated to available data; that is, the parameters are altered until model predictions approximate the information on the variables being modeled.

As pointed out by Rose et al. (1991), the approach used for configuring and calibrating the models can yield input values with unaccounted uncertainties and biases that can affect model predictions. The calibration process requires skill, especially with complex models, and different investigators may derive quite different model parameterizations. Indeed, it may not be possible to "separate the modelers and their decisions from the models and their formulations" (Rose et al., 1991). One method to assess a model is called model validation, or credibility.

3.5. Credibility (Model Validation)

Models designed to mimic exposure or responses of systems to stress require an assessment of the credibility or validity of the model. Mankin et al. (1977) and Shugart (1984) divide model testing into two basic types of procedures—verification and validation—and view model application as a measure of a model's usefulness. A model is verified by testing whether it can be made consistent with some set of observations. In validation procedures, a model is tested on its agreement with a set of observations independent of those used to estimate its parameters.

Often when testing a simulation model (or other model), it is important to ensure that the model can simulate or mimic the pattern of the system under the constraint that all parameters in the model are realistic (Shugart, 1984). The model should be able to duplicate, for example, the pattern observed in forest plantations with a fair degree of accuracy using reasonable parameters. Even though they do not use data that are strictly independent of the model, successful verifications still represent reasonable tests of the model. It should be clear from the definitions that model verification

and model validation can be the same sorts of tests. Indeed, most models are tested in the verification mode because validation data are so difficult to obtain. The drawback to model verification is that the "goodness of fit" of the model is often inflated and represents a best-case scenario. Because the same data are used to fit and assess the model, the goodness of the fit has been maximized. Thus one would not expect the same level of accuracy if the model were applied to a new system or at a later time. Model validation procedures call for independent data and estimates of how well the model will perform under more typical situations. A useful discussion of the aspects of validation are provided by O'Keefe et al. (1987).

The analysis of a model's credibility is useful not only for computer simulation models but for other models as well, such as the use of surrogate test systems (e.g., single-species toxicity test, mesocosm) for ecosystems. How well the test system mimics the ecosystem should be evaluated using validation methods. Most evaluations of test surrogates use verification rather than validation data in assessment of the test performance; however, the results from such evaluations are often misinterpreted in terms of the test system's ability to predict the effects of stress on the ecosystem (Cairns and Smith, 1989).

Model verification and validation as described here are associated with the correspondence between the system modeled and the model. Thus verification and validation do not establish truth about the model but rather an association—a point that has been made by a number of authors, most recently by Oreskes et al. (1994). This limitation does not mean that the process of model confirmation should be discarded. Rather, it means that care is required in making claims about the model. As pointed out by Oreskes et al. (1994), "the burden is on the modeler to demonstrate the degree of correspondence between the model and the material world it seeks to represent and to delineate the limits of that correspondence." The degree of correspondence is of interest in terms of a model's credibility as discussed here. Methods for analyzing uncertainty or for evaluating the limits of correspondence are described in the next section.

4. RISK CHARACTERIZATION

Models of risk and its characterization may be qualitative or quantitative. While risk models are typically quantitative, qualitative models may arise as a result of the complexity of the assessment and the lack of knowledge. Qualitative models often can provide road maps for assessing the importance of various components and pathways of a system and how a stressor affects these pathways, while quantitative models attempt to numerically evaluate risk. Regardless of the model, risk is typically viewed in terms of probability of effect given exposure or probability times consequence of exposure (Suter, 1993). For example, Suter et al. (1986) define the risk associated with toxicity to chemicals as the probability that the expected environmental concentration of a chemical exceeds the surrogate endpoint concentration (usually determined from a toxicological study). The probabilistic interpretation results from the recognition of stochasticity in the estimation of exposure and/or effect concentrations. Although estimated as a probability, risk may be represented as a distribution or a cumulative distribution function, or as a function of the exposure or the effect. When uncertainty is taken into account, the risk is no longer viewed as a single number or distribution but rather as a series of distributions (Helton, 1993). How these distributions are estimated depends on the model used to evaluate risk and the amount of information available to assess uncertainty.

The assessment of uncertainty may be qualitative or quantitative, just as the models of risk may be qualitative or quantitative. Uncertainty in the characterization of risk has to be evaluated relative to the goals of the risk assessment and the information available. Most assessment problems are not solved by a quantitative or qualitative model. Rather, it is the model that is solved and the solution is related to the problem. Thus the results of the model should be interpreted relative to the uncertainty of the model and assessment goals. The value of uncertainty analysis is realized in terms of questions such as: Can we do better given the understanding that we have of the system? What are the major sources of uncertainty and can these uncertainties be reduced? Under what situations will estimates of risk be good or bad? What are the implications that a given policy will have on management goals? Such questions can be addressed through the qualitative and quantitative assessment of uncertainty.

A selection of methods for assessing uncertainty are described below. These include the qualitative assessment of uncertainty and a variety of methods for quantitative uncertainty estimation. The methods addressed estimate properties of the distribution of risk (e.g., 95th percentile, safety factors) or the distribution function (e.g., Monte Carlo methods).

4.1. Qualitative Characterization of Uncertainty

Qualitative characterization of uncertainty employs subjective methods to assess uncertainty. These methods may be as simple as lists of the uncertainties involved in the risk assessment. The lists need not numerically evaluate the uncertainties but may provide a rough assessment of the magnitude of uncertainty (as in table 8-1). Expert opinion also provides a qualitative assessment of uncertainty. Although expert opinion is a necessary component of assessment, since experts do not agree, their opinions should be used in conjunction with data. Studies in engineering indicate a wide variation in expert opinion (Morgan et al., 1985). One would expect an even wider range of opinion in ecological assessments where problems are less well investigated (Suter, 1993, page 41).

Qualitative assessment of uncertainty may be addressed through the use of qualitative models. Simple models address only the cause-and-effect relationships, importance of factors, and the effects of possible actions. Methods for building these models include the Leopold matrix and various simulation programs (see, e.g., Holling, 1978). Qualitative models can be quite useful for sizing problems and solutions, which involves finding the right variables and factors to consider, the proper indicators to investigate, and the choice of actions to consider. Qualitative models are useful for getting "quick looks" at the problem, for investigating possible scenarios quickly, and for indicating where the knowledge uncertainties are. Although it is possible to get a rough indication of the magnitude of uncertainty with a qualitative model, qualitative models suffer from the lack of a measure of belief (certainty or uncertainty) in the results because uncertainties related to parameterization and stochasticity are hard to include in the models (Holling, 1978).

4.2. Quantitative Assessments of Uncertainty

Since in many risk analyses a model is chosen and then used to produce an estimate of risk or effects associated with a stressor, it is possible to produce quantitative estimates of uncertainty. Numerical estimates of uncertainty are primarily available for statistical and computer models. With statistical models, the measure of uncertainty is usually associated with the estimate of variance. With computer models, the quantitative analysis of uncertainty has several names (e.g., uncertainty analysis, error analysis, Monte Carlo analysis, sensitivity analysis) and may produce a single number with which to estimate uncertainty or a distribution of output that provides information on the range and magnitude of uncertainty.

From a modeling perspective, three sources of uncertainty are of interest: structural, parameter, and natural variation. Structural uncertainty analysis focuses on how different mathematical

formulations of the model alter the output and decision; parameter uncertainty analysis addresses how uncertainty in model constants change the output; and analysis of the effects of natural variation indicates how stochastic uncertainty alters the output and decisions.

4.2.1. *Uncertainty in Structure*

Uncertainty in the modeling process arises from the selection or development of the model. In an ideal application, it would be possible to have enough information to develop a model and then evaluate the ability of the model to predict ecological effects. It is more often the case, however, that the risk evaluation is complex and information is sparse relative to the complexity. This leads to a large number of possible models; hence, comparison of the models is important. One approach is to compare models by varying the components of the model, which will give rise to an estimation of uncertainty attributable to changing the structure of the model. Components that greatly alter the model output require care in evaluation and calibration.

In applying models to ecological risk assessments, a researcher may have to choose among competing models that involve different assumptions, parameter sets, spatial and temporal scales, and inputs. Typically, extensive data sets are not available to choose from among the different models based on calibration and testing. A recent technique called input mapping establishes a set of rules for comparing models (Rose et al., 1991). Given a common set of data, input mapping is a method for transforming this common set into input sets for the different models. The approach removes some of the bias arising from the need to calibrate the models for specific applications.

4.2.2. *Uncertainty in Statistical Models*

Uncertainty in statistical models is usually represented by prediction of parameter/ endpoint variance, confidence, and power. The variance is a measure of the uncertainty introduced by using a sample of observations and a model to estimate the quantity of interest. Thus variance is a measure of how far apart (i.e., how spread apart) repeated values would be if sampling could be repeated. The variance may be attributable to particular factors (e.g., sampling errors, the sampling design, and errors in handling and processing measurements) and basically represents any differences in observations or predictions that cannot be explained by the model. Although the estimation of variance is often a simple matter, especially when replicate observations are available, variance estimation can be quite complicated, requiring advanced techniques or computer simulation to obtain valid estimates (see, e.g., Pelletier and Gros, 1991).

Another measure of uncertainty in risk assessment is the confidence interval on an estimated parameter or prediction interval on a model prediction. These intervals represent ranges of values that are plausible given the variation in the data used to estimate the parameter or prediction. Parameter estimates or model predictions with greater uncertainty will have larger variances and wider confidence intervals. The confidence view is often appropriate in the development of monitoring programs (e.g., the Environmental Monitoring and Assessment Program [EMAP] and the National Acid Precipitation Assessment Program [NAPAP]). For example, as described in Thornton (1993), one interest in the NAPAP program was in determining the proportion of aquatic resources that were acidic or potentially susceptible to acidic deposition. A statistically designed survey of aquatic systems was conducted between 1984 and 1986 to provide estimates of important endpoints (i.e., number, length, area, and location of acidic or low-pH lakes or streams) based on a samples of lakes and streams. The survey was designed to produce estimates of endpoints for which the researchers could have high confidence that the estimates would be precise (i.e., have a low variance or produce a narrow range for the estimate of the endpoint). This confidence was accomplished through the development of a statistical design to reduce bias and through the choice of an adequate sample size. The desired precision is a choice made by the risk manager and assessor prior to the study and is determined in part by the cost of sampling and policy implications (i.e., how important is it to have good estimates of the endpoints).

In a statistical analysis of information, power analysis is a method for quantifying uncertainty in a statistical test. In this context, the term power is used to mean the chance that a false statement is rejected. For example, if one is using data to determine if a potential stressor affects an ecosystem and it actually does, then the power of the test is a measure of how likely it is that one will decide that the stressor actually has an effect. The power of a test is a number between zero and one: Values of power near one indicate that the test is extremely sensitive, while values near zero indicate little sensitivity. Power is limited by the number of observations used in the test (more is better), the variability in the system (less is better), the probability of falsely rejecting the null hypothesis when it is true (higher probability results in higher power), and the size of the suspected effect (a greater effect is more likely to be detected).

Power analysis can be helpful for the decision-making process in risk assessment. Because power involves sample size, ecological effect size, and variability, an analysis of power can help answer such questions as: How much information is required to be fairly certain that an important ecological effect can be detected? Given a fixed cost for an assessment, how great an ecological

effect can be detected with high certainty? and, If a study is already completed, how much uncertainty is there that an ecological effect of a given size was not detected?

An interesting application of error rates and statistical uncertainty analysis is found in the case study on Commencement Bay (Cirone and Pastorak, 1993). In this study, the apparent effects threshold (AET) was developed to estimate a threshold concentration above which biological effects would be expected. The AET may be viewed as a "critical value" in a statistical test, and the process has error rates that affect reliability. The reliability of this measure was evaluated by applying the process to an independent data set. The evaluation indicated that the method had high reliability in predicting biological effects (high power or low Type II error rate) and a low rate of false positives (low Type I error rate).

In some cases, variance represents all or most of the uncertainty in the risk analysis process. However, in other cases (e.g., when data are available at one site but the assessment is carried out at another site), additional sources of uncertainty are likely to be present and use of an estimated variance may result in overconfidence in results. For such cases, some understanding of the other sources of uncertainties (e.g., differences between sites) is necessary. The additional uncertainty is sometimes represented as model "bias," which refers to the deviation of the predicted or estimated quantity from the true value. Note that one difference between variance and bias is that variance typically is calculated assuming the model is true when calculating error, while the bias is calculated using the true model. Bias (squared) and variance are sometimes combined to form an overall measure of uncertainty called mean squared error. Bias, variance, and mean squared error are often used to evaluate the prediction capability of models (Bartell et al., 1986).

4.2.3. *Uncertainty Factors/Extrapolation Models*

One approach to incorporating uncertainty into a risk analysis of toxicants is through an uncertainty factor. This approach is most often applied when the result of the risk analysis is a single number, such as the amount of a pesticide or chemical that is allowed into a system. The approach may be quite simple. For example, one uses the best available information to pick a number, then the number is divided by a safety or extrapolation factor (e.g., 10, 100), which indicates the degree of magnitude of uncertainty in extrapolation from laboratory or test results to the environment. Perhaps the most common example of an uncertainty factor involves the quotient method (Barnthouse et al., 1982), which computes the ratio of the environmental concentration that is expected by the test endpoint concentration (e.g., LC_{50}). The ratio is then multiplied by a risk factor (or an assessment

factor) and compared to 1.0. Values less than 1.0 indicate safety. Bioaccumulation factors and acute/chronic ratios also may be viewed as uncertainty factors.

A more recent approach developed to account for uncertainty in choosing a safe level of a stressor is called the extrapolation method. In this approach, the unit to extrapolate over is defined. For example, to set a level of safety for a chemical, tests on single species may be used with an interest in protecting all the species in an ecosystem. Thus, since one would use single-species results to extrapolate across species, the unit is the species. Each unit is tested separately and the test result (e.g., an LC_{50} value) is viewed as a number representing a sample from the population of units. By relating the population of units to a distribution of values, statistical theory can be used to estimate a number that is smaller than most of the values in the population (e.g., smaller than 95 percent of the values) for most sets of test units of the same sample size (e.g., 95 percent). Theoretically the method is better than the simple approach of choosing an uncertainty factor, since the uncertainty factor applied is based on the amount of data used to assess the stressor effects and the variation in response. The method is being used currently to set safety levels for new chemicals in the European Community (van Leeuwen, 1990), and a similar model is being used by the U.S. Environmental Protection Agency (EPA) (Stephan et al., 1985). The approach does not include all potential sources of uncertainty, however, and may be subject to biases (for a more extensive summary along with references, see Suter, 1993).

Uncertainty factors can be made more precise when data relating different scales or endpoints are available. For example, if a chemical model is used to estimate the toxicity of a new chemical, the relationship between other chemicals and toxicity can be used to predict the toxicity of the new compound. Uncertainty then is estimated based on the variance of the prediction (Sloof et al., 1986). This method is useful for the evaluation of uncertainty in the toxicity of chemicals, requires a reasonably large data base, and works best for chemicals of similar characteristics (Lindgren et al., 1991). An example in which this approach is used to estimate uncertainty attributable to extrapolation across different levels of complexity is described by Barnthouse et al. (1990) and is called the analysis of extrapolation error (AEE). The method uses information on a well-studied endpoint (e.g., rainbow trout LC_{50}) to predict the distribution of a desired endpoint (e.g., production of juvenile brook trout), using data available on different stressors (usually chemicals). The prediction is usually carried out using regression models and may extend over several levels of extrapolation, based on available data. The analysis ends by estimating the probability of risk by computing the probability that exposure exceeds the effect endpoint. Although the approach usually assumes that the endpoints

follow a normal or lognormal distribution, other statistical models may be used. Details and examples are given in Suter (1993).

4.2.4. *Uncertainty in Computer Models*

Uncertainty in computer models is usually approached through the analysis of errors in parameters and analysis of the effects of stochasticity. Uncertainty assessment through error analysis yields estimates of model output variance as related to uncertainty in parameter estimates. An additional uncertainty factor may be added to account for variance related to the stochasticity of the system, and although it is not a common practice, one also can estimate the uncertainty or bias attributable to the incorrect model. Most estimates of uncertainty are obtained by using computer simulation methods (i.e., Monte Carlo analysis), although the variance of the model also can be estimated using "first-order" analysis or sensitivity analysis. This method, which uses small perturbations or partial derivatives to estimate model variance attributable to parameter imprecision, is best suited to simple models, linear models, and models with small variances (Gardner et al., 1981; Pelletier and Gros, 1991).

The techniques used to distinguish how model responses are conditioned by inputs and by model parameter values are generally referred to as "uncertainty analyses" (for a review, see Beck, 1987). The technique used to conduct a sensitivity analysis of linear models is one of the classical approaches to the problem (Tomovic, 1964). Because of nonlinearities and stochastic elements, most of the techniques that are widely used in ecological sciences employ Monte Carlo simulations, although first-order approximations quite often are useful. O'Neill and Gardner and their colleagues have been particularly productive in the investigation of ecosystem models using these methods (e.g., O'Neill and Gardner, 1980; Gardner and O'Neill et al., 1981, 1983).

One also can examine variation about a nominal system trajectory, which reflects uncertainty in model parameters and in the specification of driving variables. Such analysis is similar to what has been referred to as "error propagation" (e.g., Garen and Burges, 1981; Guymon et al., 1981). For a suite of model trajectories representing a spatial environmental gradient, appropriate error propagation statistics can be computed (Gardner and O'Neill, 1983). These methods allow a direct assessment of transient error variance, which may be of considerable importance when the models are run in a forecast mode, as would be the case, for example, in an environmental assessment.

4.2.5. *Sensitivity Analysis*

Sensitivity analysis is a mathematical technique in which the partial derivatives of the model output variable(s) are taken with respect to the parameters to measure how much the model output changes when the parameter is changed by a small amount. Parameters with high sensitivity are important in the model. The value of a sensitivity analysis is that sensitivities of the parameters can be used to rank the parameters in terms of importance. This information can be used to quantify uncertainty (using a first-order variance approximation), to indicate where additional research is required (identifying which parameters need to be well estimated), or to develop a reduced set of parameters for further study.

Although the technique sounds simple, a sensitivity analysis can be complicated by a number of factors. For example, the approach, as described above, is of limited use when the model is highly nonlinear or if there are strong relationships between parameters. Also, sensitivities may not be constant over time or space. Both of these problems hinder the ranking of variables in terms of importance. Methods to deal with nonlinearities and relationships between parameters are generally complicated (Downing et al., 1985); some of these are described below. Bartell et al. (1992, ch. 6) gives an example of a sensitivity analysis involving time.

4.2.6. *Monte Carlo or Error Analysis*

Monte Carlo uncertainty analysis, or error analysis (Gardner et al., 1980), is another method that attempts to assess the importance of parameter uncertainty. Parameters are viewed as random quantities, and a distribution is assumed for each parameter or a multivariate distribution is assumed for the set of parameters. Random values are drawn from the distribution(s) and the values are substituted for the model parameters. The values of the input parameters and model output are then recorded. The process is repeated numerous times (more than the number of parameters; preferably enough times to get a stable estimate of effects) to produce a large set of model parameters and output. Correlation methods then are used to relate changes in the parameter with the output of the model. High (positive or negative) correlations indicate that a parameter is important (see, e.g., Bartell et al., 1992). Variations on this approach depend on the complexity of the model, the number of parameters that need to be assessed, the approach to assessing correlation, and the method by which the potential parameter values are constrained. This same process may be applied to assess sensitivity, by restricting the variance so that it is small (1 to 2 percent of the mean).

The current view is that the Monte Carlo analysis is the better approach for assessing parameter uncertainty. Sensitivity analysis focuses on the effects of small changes in parameter values, and (reasonably) a linear relationship between the model variable and the parameter is needed for accurate results. When the relationship has higher order effects, the method will produce results quite different from the Monte Carlo method. Since most models have nonlinear components, the Monte Carlo approach is preferred.

Difficulties can arise in using the Monte Carlo method when the number of parameters is large and little is known about the variation and distribution of the estimated parameters. When the computation burden is large (as with a large number of model parameters or an expensive, time-consuming model) computational schemes are required to produce a good analysis. An adaptation of the above procedure developed by Iman and Conover (1979, 1982) can be used to gauge model response over a wide range of variation. The method has been successfully used by Jaffe and Parker (1984) with a stream pollution model and by Wolock et al. (1986) with a hydrological model. The latter adaptation of the procedure consists of two major steps (Jaffe and Ferrara, 1984). First, the models are exercised in a Monte Carlo mode using a Latin hypercube sampling design (a design to pick values from parameter sets of equal probability). The model parameters and the parameters describing the input are ranked, as are the model responses of interest. Second, the ranks of the responses are regressed on the ranks of the parameters and the usual statistics of multiple regression are used to infer the relative importance of each of the parameters on the responses.

Another approach, which is applied when there is a large number of parameters, involves approximating the model with a polynomial model or response surface. First, the important parameters are selected using sensitivity analysis. Then they are perturbed and a response model is fitted as a function of the parameters and the model output. The response surface is then analyzed (Cox and Baybutt, 1981).

A number of difficulties with the analysis of uncertainties in computer models remain unresolved. With complex computer models it may not be possible to estimate accurately the distribution or variance of estimated parameters or to estimate the covariance between two or more estimated parameters. If parameter estimates are obtained from literature sources, little information may be available about the distribution of these estimates; often there may only be an upper and lower bound on possible estimates. The use of a complex scheme (e.g., Latin hypercube sampling, response surface analysis) may add other uncertainties and modeler bias. Complex models can require a long time to run, even on high-speed computers, and shortcuts to assessments may miss information

associated with certain combinations of parameters. Models may not converge for all sets of parameters or may converge to absurd solutions.

Uncertainty in computer models attributable to parameter uncertainty can be expressed in terms of effects on model output or lead to a rank ordering of parameters. The former provides a quantitative estimate of the effect of the parameter, while the latter provides an indication of the relative effect. The effects of uncertainty can be combined to assess the effects, for example, of subsets of parameters, components of the model, and direct versus indirect effects (Gardner et al., 1980).

4.2.7. *Stochasticity in Models*

Stochasticity in models can be incorporated through the use of a stochastic model of the ecological endpoints or by adding stochasticity to the risk model. One approach is through the use of stochastic models, which are based on stochastic differential equations rather than deterministic differential equations (Tiwari, 1979; Tiwari and Hobbie, 1976), and treat the endpoint as a random variable. The difficulty with stochastic differential equation models is the analytical complexity.

Another approach, common to computer models, is to incorporate uncertainty from natural variation into the model to make a stochastic simulation model. Distributions are assumed or estimated from available data for input variables (e.g., weather, temperature). Random samples are then selected from these distributions and used to generate one model output. By repeating the process, a scenario of possible output is generated. This scenario is usually summarized by a distribution of output values or by the mean or variance of the output.

Stochastic simulation models can be quite useful for assessing the importance of the stochastic component of uncertainty and assigning limits to the sensitivity of the risk estimates. Given information on natural variability, the severity of an ecological insult can be assessed as a function of the risk. When the natural variation is great, the estimate of risk is also likely to be highly variable and the variability of the risk estimate will only improve with increased stress. Natural variability limits our ability to separate the stressor's effects from naturally occurring effects. Understanding natural variation, however, may lead to a better understanding of when and where the stressor is most likely to have the greatest impact.

Some knowledge of the effects of stochasticity is of value in planning and interpreting risk assessments. In systems with greater natural variability, the amount of variability may determine whether additional studies are worth undertaking. Knowledge of the variability of endpoints may be

quite useful for choosing among various endpoints. For instance, endpoints with a high "signal to noise ratio" are valuable for assessing effects. Thus criteria may not be only whether the particular endpoint represents something of value but also whether it is robust to the environment yet sensitive to the stressor.

Parameter error analysis and stochasticity are often addressed in the same manner—through Monte Carlo simulation methods. Although the analyses of these as aspects of uncertainty are similar, the methods differ in purpose. The focus in error analysis is on how uncertainty in parameter estimates affects prediction. An error analysis may indicate which parameters are influential or may indicate the magnitude of uncertainty in the analysis. The aim in introducing stochasticity into a model is to produce a range of predicted values rather than a single value. The range of values may indicate, for example, how predictions of stressor effects vary across space or over time. Combined, the analyses are useful for obtaining an overall idea of variability in the output resulting from lack of certainty in parameter estimates and from variation in inputs.

The information from the analysis of stochasticity and/or parameter uncertainty also can be used to describe risk as an expected value (mean or median) and uncertainty bounds, a distribution rather than a single number or even a distribution with bounds. Since the expected effects of a toxicant or other stressor are influenced by stochasticity, site selection, and physical and chemical factors, a distribution of risk may better represent the expected range of risk that may be encountered in the field.

Other approaches to quantifying uncertainty are available but have not been extensively applied to ecological risk assessment problems. These include "belief functions," fuzzy mathematics, dependency bounds analysis, and Bayesian methodologies (see, e.g., the papers in von Furstenberg, 1990; Ferson and Long, 1994).

An example of the use of uncertainty in simulation models is described in a case study on acidic deposition (Thornton, 1993). In this study, a watershed chemistry model was developed as a predictive exposure-effects model. The model was calibrated on five watersheds and confirmed by comparing model output with external lake and stream data. Also, sensitivity analysis was used to identify parameters that required specific attention during the calibration process. The model was used to predict future changes in acidification for individual lakes and regions and to compare the effect of different sulfur reduction scenarios on the proportion of acidic systems that recover and proportion of systems that become acidic in the future (50 years).

Model prediction uncertainty was evaluated in terms of sampling variability, model input, and calibration error. The analysis of sampling variability was carried out by varying the input proportion of lakes that were acidic. Model input effects were evaluated using the Monte Carlo method in which model parameters were randomly selected from distributions and chosen for the parameters, and then new predictions were generated. The uncertainty analysis was used to generate an uncertainty range for the change in the acid-neutralizing capacity of the water systems. Risk was summarized as the median change in acid-neutralizing capacity and in the percentage of systems falling within a given criteria, with error bounds. Because of high uncertainties, these results could not be used to make strict probability statements; rather, they were thought useful for making relative comparisons. For example, a use of the analysis was to compare the watershed-to-watershed variability with the uncertainty generated from model input. It was found that input uncertainty is generally smaller than variability between watersheds. This result is useful for deciding on the scale at which the models should be applied and where data should be collected.

4.3. Reducing Uncertainty

In some ecological risk assessments it may not be possible to reduce uncertainty enough to allow reasonable risk estimates to be made. Nonetheless, in many risk assessments, good scientific practices will reduce uncertainty. These include adherence to quality assurance methods, use of statistical principles in the design of studies, use of auxiliary information, and use of multiple lines of evidence.

In assessments where data are to be collected, the use of data quality assurance methods can avoid uncertainties (Liabastre et al., 1992). These methods focus on the use of sound practices in the design of data collection objectives and sampling and analytical plans, in the execution of the sampling plans, and in the assessment of data. Attention to the principles of quality assurance can reduce errors attributable to the mishandling of samples as well as poor laboratory or field practices and can provide consistency in the collection and verification of data.

Moreover, when data are to be collected it is important to use statistical principles to reduce uncertainty. Statistical approaches to data collection are based on replication, randomization, and control of variation (i.e., blocking or stratification). Replication suggests that repetition of observations are made on the stressor. Care must be taken that the repetition represents as close a new measurement as possible and not a "pseudoreplicate." Randomization implies that the measurements are taken in a probabilistic fashion to avoid bias induced by order effects. Finally, part

of statistical methodology is concerned with methods for controlling variation in studies. Some methods include forming blocks or strata of units before applying treatments or taking measurements, and creating composites of material before taking measurements or accounting for variation attributable to factors that are influential but not of interest (i.e., covariates).

When the model involves different spatial units, it may be of value to subdivide spatially or temporally or use additional information to improve prediction. Hettelingh et al. (1992) show that when parameter values in an acidification model are split according to the sensitivity of the region that the model is applied over, explanation of the variance in predictions by model parameters can be improved. Thus it is important to identify relevant strata in ecosystems and see how the model varies over these strata.

Another approach to reducing uncertainty is to use multiple lines of evidence. Combining a field study with a laboratory study can be quite powerful for eliminating uncertainty. The field study may be useful for indicating an association between the stressor and the ecosystem, while the laboratory study can establish a causal relationship. Little work seems to have been done on assessing uncertainties for such combined studies.

5. DESCRIBING UNCERTAINTY

An important element of uncertainty in risk analysis is the communication between the risk assessor and the risk manager. While it is the job of the risk assessor to produce the best estimate of risk and to defend that estimate, it is also important that the uncertainties associated with that estimate be explained. As indicated in the case study on acidic deposition (Thornton, 1993), two important problems identified in the study were that quantitative estimates of all components of uncertainty must be presented clearly to decision makers and better procedures are needed for describing uncertainty and communicating risk. Indeed, it is the role of the risk manager to insist that uncertainties be explained and the role of the risk analyst to invest time to ensure that the methods and results are presentable. Issues that need to be considered are how the uncertainty will be described (qualitatively or quantitatively), what information needs to be displayed, and what is the best way to display the information. In quantitative assessments, another consideration is how to convey information about parameter sensitivities, stochasticity, and model uncertainty (Morgan and Henrion, 1990). The meaning of uncertainty, as estimated from computer models, requires special attention.

The explanation of uncertainty may involve qualitative or quantitative components. The qualitative components may involve other pathways to models, assumptions about the important endpoints to measure, or assumptions about exposure. Other important elements would include information about the importance of various components. For example, rank orderings of parameters may be quite useful for indicating not only which parameters have the highest uncertainty but also which parameters have the highest certainty. Grouping the parameters in terms of sensitivity may be valuable for deciding what information is needed. Such a list can help direct future studies intended to reduce the uncertainty of the estimated parameters. We recommend that this list or set of explanations and concerns be included in a separate section in the risk document or in subsections associated with the different phases of the Framework. Some, but not all, of the issues that must be addressed include: What led to the choice of endpoints and what are the consequences of this choice? What are the limitations of existing data and how do they affect the assessment? Why were particular models chosen? Which components of models are well studied and which are not? What are the effects and limitations of extrapolations? Additionally, the issues in the model conceptualization process and in the building of models for exposure and effects assessments need to be described in the risk document. It is advisable to have the document reviewed both internally and externally for possibilities not considered by the risk assessor.

Quantitative aspects of uncertainty need to be described in terms of variability concerning the risk estimate or in terms of effects of parameter uncertainty and stochasticity on ecological models. We recommend that risk and uncertainty be represented as distributions and families of distributions when possible, rather than as a single risk estimate. Such representations can be presented graphically by drawing density functions or cumulative distribution functions. A cumulative distribution is useful because it is easily interpreted as the probability of exceeding a level (e.g., as the proportion of sites exceeding a "no effect" level). This approach also provides a range of risk as well as a central measure of risk. Additionally, the properties of this distribution can be summarized.

Further, since it is often worthwhile to consider a variety of scenarios in evaluation of risk, the influence of weather might be included in a study by considering effects of high stochasticity versus low stochasticity. The range produced might indicate when certain policies will be most or least effective. There is no "best" approach to producing these distributions since methods vary based on the complexity of the model and information that is available. We further recommend that sensitivity analysis be used where possible to assess important components of the model(s) and advise that error analysis be used in the assessment of effects of uncertainties.

Understanding the relationship between environmental variation or stochasticity and risk evaluation is important because it indicates the limits of the predictive ability of the procedure. With extreme natural variability, a simple numerical risk estimate cannot be produced with high confidence of correctness. The risk manager has to be made aware that the predictive effects of a stressor have to be evaluated relative to natural variability and that as the background variability increases, the predictive ability decreases. Natural variability is especially important in selection of endpoints, since endpoints with high response ability also may be responsive to natural changes in the system. In the best case, the risk assessor will be able to indicate whether the process can be improved and will set limits for that improvement.

The possibility of high variability in ecosystem risk studies suggests that the approach to risk management must be able to adapt to changes in certainty. As knowledge increases about a stressor, hypotheses about causes of effects can be eliminated and models of risk and the risk process improved. This adaptive approach (Holling, 1978) to assessment and management of risk is necessary in the face of the high uncertainty of many ecological assessments.

6. SUMMARY

Uncertainty analysis is an essential component of the ecological risk assessment process. Thus it is particularly important that the uncertainties associated with the components of the assessment be described and when possible quantified so that these uncertainties and their implications are conveyed to the risk manager.

Uncertainty analysis may be quantitative or qualitative. Often uncertainties associated with the components of the assessment can be analyzed in a quantitative manner, but it is difficult to combine the information into an overall quantitative evaluation of uncertainty. At some point in the risk assessment a decision must be made regarding the detail of the uncertainty assessment. This decision depends not only on available information, but also on the magnitude of the hazard and the ecological effect.

This chapter does not attempt to provide an exhaustive treatment of the sources or methods for evaluating uncertainty. Rather, uncertainties associated with more commonly used methods for evaluating risk are described, and some of the more standard tools for assessing the magnitude of uncertainty are presented.

In the evaluation of uncertainty, one must keep in mind the words of George Box (1979): "all models are wrong but some are useful." Uncertainty analysis is the process of determining the usefulness of models, of assessing the limitations, and of reporting on those limitations. The analysis forms an important part of risk assessment of ecosystems and needs to be a component of the process.

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Issue Paper
on

RISK INTEGRATION METHODS

Richard G. Wiegert
Institute of Ecology
University of Georgia
Athens, Georgia

Steven M. Bartell
Center for Risk Analysis
SENES Oak Ridge, Inc.
Oak Ridge, Tennessee

Prepared for:

Risk Assessment Forum
U.S. Environmental Protection Agency

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Risk characterization is the interactive process of extracting and integrating decision-relevant information from hazard, dose-response, and exposure evaluations and rendering it comprehensible to a diversity of users.

AIHC, 1992

1. INTRODUCTION

This chapter discusses characterization of ecological risks in relation to the *Framework for Ecological Risk Assessment* (U.S. EPA, 1992). In doing so, the chapter evaluates the range of approaches available to the risk assessor for integrating data on exposure and stressor-response profiles, and it discusses the methodology available to implement these approaches. The exposure profile summarizes the magnitude and spatial-temporal patterns of exposure. The stressor-response profile summarizes data on the effects of the stressor (i.e., its relationship to the assessment endpoint).

The *Framework Report* identifies several approaches that can be used to characterize ecological risk: (1) comparing single-effect and exposure values; (2) comparing distributions of different exposures and effects on the same system; and (3) examining different exposures and effects in different systems or considering multiple effects from single exposures on the same or on different systems. Methods suited to evaluations using the above approaches involve some form of physical or mathematical modeling. This chapter addresses the applicability of these methods and approaches, and it discusses their advantages and disadvantages. Where possible, the critical assumptions underlying the use and interpretation of these methods are stated.

Also, techniques available for translating the exposure profile and the stressor profile into estimates of ecological risk are presented. The principal methods discussed include the use of several different empirical models, process (or mechanistic) models, experimental facilities (e.g., microcosms, mesocosms), and, finally, whole-system manipulations. The importance of follow-up monitoring studies is addressed, as is the relevance of ecological risk characterization to the U.S. Environmental Protection Agency's (EPA's) Environmental Monitoring and Assessment Program (EMAP).

This treatment of risk characterization also provides example applications, outlines the information requirements for implementing the various methods, and points out the strengths and limitations of these separate approaches. Issues of uncertainty in models and data are discussed in relation to characterizing risk and recovery. Finally, risk characterization is considered in the context of the overall ecological risk summary.

2. PREVIOUS CONSIDERATIONS OF RISK CHARACTERIZATION

Risk characterization involves integrating all information gathered and analyses performed in the course of the assessment and communicating it to risk managers in an understandable manner. Not surprisingly then, the process of risk characterization continues to be a focal point for improving risk assessment and risk management (see, e.g., Paustenbach, 1991; AIHC, 1992). Previous considerations have produced general points of agreement concerning the necessary components of an effective risk characterization. In particular, important aspects of risk characterization were recognized at the American Industrial Health Council's 1991 workshop titled "Improving Risk Characterization"; these are outlined in table 9-1. The AIHC emphasizes two requirements for full characterization of risk: (1) a full description of all the qualitative and quantitative elements of the risk analysis must be provided, and (2) a candid discussion of the uncertainties associated with all components of the risk analysis and their implications for the overall assessment is critical to full characterization of risk. A complete summary of risk estimates, assumptions, limitations, and uncertainties also is recognized by Paustenbach (1991) as fundamental to the risk characterization process.

Table 9-1. Improving the Technical Content of Risk Characterization (AIHC, 1992)

Relevance	Ensure compatibility between the needs of the assessment and the nature of the technical results
Clarity	Highlight the important results, remove extraneous information, make ample use of tables and graphs
Balance	Realistically portray the range of scientific views, provide the basis for assumptions and judgments, acknowledge the uncertainties
Consistency	Be consistent in terminology, definitions, and the processing of information from one characterization to another
Detail	Provide the necessary level of detail appropriate for the end-users of the risk characterization

3. RISK CHARACTERIZATION AND THE FRAMEWORK

Because the risk characterization process represents the integration of all aspects of the framework, each of the major components is reviewed briefly from the perspective of its potential contribution to risk characterization.

3.1. Problem Formulation

Problem formulation constitutes the first step in the assessment framework. During this initial phase, the entire risk assessment team, including managers and assessors, determines the scope, goal(s), resources, and possible analyses that will constitute the overall assessment.

The scope addresses the level of biological organization (e.g., individual endangered species, population, community, landscape, region/watershed/drainage basin), coupled with the degree to which the physicochemical environment is involved. The scope of the study may be determined largely by the nature of the stressor. For example, will the assessment consider risks associated with chemical (toxicants, nutrients), physical (habitat change), biological (disease), ecological (introduced exotic competitors), or mixed stressors? The scope also may be determined by the underlying motivation for the assessment. For example, will the assessment be used by regulators to address specific environmental legislation (e.g., Resource Conservation and Recovery Act [RCRA]/Comprehensive Environmental Response, Compensation, and Liability Act [CERCLA]), or will it be used by the private sector in planning new products that minimize future environmental impacts and liability?

The nature of the stressor will identify in large part the relevant scale (temporal or spatial) of the assessment, and, therefore, relevant scales to be considered in choosing methods for risk integration. Examples of different scales are ponds, streams, lakes in the spatial domain, and single chemical spills versus periodic (e.g., agricultural pesticide application) or continuous (e.g., atmospheric acidic deposition) events in the time domain. Spatial scales also might range from the local impacts of spills to landscape and regional scales (e.g., acid deposition, pest outbreaks) and even to global scales (e.g., increased ultraviolet-beta radiation).

The problem formulation phase also includes inquiry concerning the nature of anticipated ecological effects in relation to the stressor. The scale, level of resolution, and available information can assist in selecting among the various approaches and tools for estimating and characterizing risk.

The goal of the risk assessment may require either predictive or retrospective types of assessments. Point estimates, estimates of ranges, or both may be necessary. Taken together, the

statements of scope and goal(s) will permit evaluation of the decision makers' requirements to (1) discuss how these requirements further constrain the studies; (2) list the degree of advance knowledge of regulatory options that may be required; and (3) consider the resources, both financial and personnel related, that may be required.

The level of reliability in the approach(es), the analyses, and the feasibility of the requirements must be evaluated. In general, these requirements can be met only by thoroughly considering the statistical techniques that are available and feasible for defining and characterizing reliability in anticipation of the quantity and quality of information or data that will be available during the assessment.

The final product of the problem formulation phase is a conceptual model of the entire assessment. The model outlines the objectives of the assessment and specifies the suspected causal relations between stressors and endpoints (e.g., an influence diagram might be an effective conceptual model). Importantly, the conceptual model remains as a working tool in an iterative approach to assessing risk. The model should identify criteria for conditional milestones at which risk management alternatives can be selected based on results of the assessment. Similarly, the model should facilitate identification of additional information needs in relation to the information's value in selecting among management alternatives.

3.2. Problem Management

Managing the problem of risk assessment depends critically on how the system at risk is viewed and what is expected of any action taken to avoid or ameliorate the risk. The insights of Holling (1986) into the operation of systems are relevant in this regard.

Holling considered the way ecosystems (of whatever complexity or spatial/temporal scale) are perceived by humans, how the systems function, and how the interaction of the perception with the function could affect attempts to manage ecosystems in the face of human impacts. He noted three ways in which ecosystems are viewed: (1) as being centered on a constant equilibrium, (2) as having multiple equilibria and operating in response to both internal and external perturbations, and (3) as being centered on a constant equilibrium with evolutionary change, both external and internal, factored in. He then proposed that ecosystems function in a series of four stages, progressing slowly from exploitation (i.e., early succession) to conservation (i.e., a temporally stable steady state or limited cycle), then ultimately through creative destruction (i.e., subjected to such agents as fire, senescence, pests) to renewal (i.e., when nutrients are accessible and energy is available), at which

point the system is ready for the cycle to be triggered again by exploiters. Attempts to manage an ecosystem by reducing variability and/or keeping the system in one stage of this cycle are fraught with difficulties. Evolution usually changes the system such that it is susceptible to progressing to the next stage of the cycle, often resulting in more variability than would be observed in a natural progression. Holling cites several examples in which human intervention aimed at suppressing pests or fires or improving fisheries resulted in short-term successes that led to worse long-term situations.

It is not difficult to imagine getting into the same difficulties by attempting to manage systems subjected to risk from the kinds of stressors considered in the framework. Thus, we would do well to keep these definitions and cautions in mind while developing assessment guidance. They have particular application to risk integration given the dependence on models and prediction.

3.3. Analysis

Analysis in regard to risk characterization involves evaluation of exposure and effects information in the context of the endpoints at risk. The data for both exposure and effects need to be compatible with the risk integration technique to be used. How does the resource at risk (e.g., an endangered species, a typical farm pond, an entire commercial fishery) or the enduse of a product dictate the availability and quality of exposure and effects information? What considerations are necessary to ensure that the exposure and effects data are compatible for use in the appropriate risk integration techniques? Answers to such questions will help formulate the approaches (i.e., methods, models, and data) used to estimate ecological risks and will help determine the architecture of the risk characterization.

4. RISK CHARACTERIZATION

This section considers potential methods and approaches for integrating exposure and effects profiles and provides estimates of ecological risk. These qualitative and quantitative estimates constitute the basis for the subsequent risk characterization. Their strengths, weaknesses, and uncertainties are added to the overall assessment.

4.1. Empirical Models

If sufficient data on the responses of any ecological system to perturbation by a stressor are available, empirically characterizing risks in relation to the assessment endpoints may be possible. A number of alternative methods exist for estimating risks in this manner. It should be recognized in further development of the framework, however, that all quantitative methods are ultimately empirical, requiring numerical data at some point to arrive at an answer. The term empirical is used in this chapter to refer to direct analysis of available data or use of strictly statistical procedures and does not include the use of process models or experimental approaches.

4.1.1. *Single-Value Comparisons*

Qualitative assessments can be performed by comparing point estimates of exposure to single measures of toxicity. The quotient method (Barnthouse et al., 1982; Suter, 1992) is based on a ratio of an exposure concentration to a toxicity benchmark, appropriately adjusted by an application factor determined by the source of the toxicity benchmark (table 9-2).

Several assumptions are explicit or implicit in the quotient method when used to assess risk. First, in routine application, exposure concentrations are assumed invariant in space or time. To the extent that the particular circumstances violate this assumption about exposure, the results of the quotient method could provide inaccurate estimates of risk. For example, Breck et al. (1988) demonstrated the implications of spatial-temporal variability in exposure toxic effects for fish populations exposed to aluminum.

Second, and perhaps most important, available effects data are assumed to be suitable for extrapolating directly to the field. Quotient estimates are based on assumptions that the species most sensitive in the laboratory is similar to the most sensitive species in the field. The application factors in table 9-3 were derived empirically to adjust for some components of the uncertainties associated with these kinds of extrapolations, based on the source of the effects data. The reliability of these values, however, has yet to be rigorously examined. The factors appear to have some empirical

Table 9-2. Methods for Error Propagation in Mathematical Models

Method	Application	Reference
Parameter perturbation	Lakes and reservoirs	Recknagel, 1984
	Biochemical oxygen demand (BOD) in rivers	Rinaldi and Soncini-Sessa, 1978
First-order variance propagation	Eutrophication model of Saginaw Bay, MI	Scavia et al., 1981
	Eutrophication model of Lake Morey, VT	Walker, 1982
	Piezometric heads in an aquifer	Dettinger and Wilson, 1981
Probability density function/moment method	BOD in Sacramento River	Tumeo and Orlob, 1989
Fourier amplitude sensitivity test (FAST)	Autocatalytic chemical reaction	McRae et al., 1982
Deterministic uncertainty analysis	Waste flow through an aquifer	Worley, 1987
Monte Carlo methods	Stream ecosystem	Gardner et al., 1981
	Anthracene in ponds	Bartell et al., 1983
	Bioenergetics of fish growth	Bartell et al., 1986a
	Eutrophication in Saginaw Bay, MI	Scavia et al., 1981

Table 9-3. Application Factors for Extrapolating to Concentrations of Concern (U.S. EPA, 1984)

Data Source	Application Factor
Single measured or estimated acute LC_{50}, EC_{50}	1,000
Lowest of five acute LC_{50} or EC_{50} values for invertebrates and fish	100
Lowest chronic No Observed Effects Concentration (NOEC) value for most sensitive species in acute tests	10
Concentration of concern determined from field measurements	1

validity (Suter, 1992); however, a process-level understanding that would suggest these factors has yet to be offered. Additionally, factors of this kind have not been developed for stressors other than toxic chemicals. Although in principle the quotient method could be developed for other disturbances, its current applications are mainly for risks posed by toxic chemicals.

The quotients appear most useful for screening purposes or for assessing comparative effects of natural or human-caused disturbances. Assessments based on a comparison of single- effects values or point estimates are not consistent with a probabilistic framework and, strictly, should not be considered as quantitative estimates of risk. Note that these quotients, while unitless, do not share the same range (i.e., [0-1]) as probabilities. Thus it remains difficult to integrate results from quotient methods with any assessment endpoints that are framed in terms of probability (e.g., the probability of a 20 percent decrease in annual production of a species of interest).

4.1.2. *Joint Distributions*

More realistic assessments can be performed by comparing distributions of exposure to distributions of toxicity data (e.g., Suter et al., 1983). Comparisons of these distributions recognize the spatial-temporal variability in exposure and the variability in ecological response.

There are several sources of variability in exposure. Point sources of pollutants can vary in time. Nonpoint sources can vary in both space and time; for example, aerial application and the subsequent runoff of pesticides from agricultural fields will show strongly seasonal and topographically related heterogeneity. These sources of variability suggest that realistic assessments or estimates of risk require that distributions of exposure be used.

Several factors contribute to variability in ecological response to disturbance. For example, a population exposed in nature will likely differ from its taxonomically similar laboratory counterpart with respect to genetics, age (size) structure, ecological context, and history. These sources of variability suggest that distributions of response should be used in comparisons of exposure to disturbances for assessments or risk estimates.

In this approach, risk is calculated as the probability that the exposure concentrations and the concentrations correlated with measurable effects represent the same underlying statistical distribution; that is, risk is the probability that the two distributions overlap perfectly. Methods in basic statistics have been developed to measure this degree of overlap. The methods have been modified to easily determine the contribution of bias and variance of each distribution to the degree of overlap (i.e., risk) (Bartell et al., 1986b).

In some applications, risk has been characterized as the degree of overlap of a distribution of exposure concentrations with a point estimate of a concentration associated with the toxic response (e.g., Suter et al., 1983). In this approach, the implied test is that the effects concentration is a statistical sample of the distribution of exposure concentrations. Thus it is a determination of the probability that the distribution of exposure concentrations includes the effects concentration. These kinds of integration methods are intermediate between single-value comparisons (i.e., the quotient method) and the comparison of joint distributions.

The comparison of joint distributions could be performed by generating a distribution of quotients through randomly sampling from the distribution of exposures and the distribution of effects concentrations and then performing the division. (The operation is defined as long as nonzero concentrations are required to measure the effect of interest, which holds by definition in a controlled toxicity experiment.)

Joint distributions also have been developed to describe expected exposure concentrations, acute toxicity data, and chronic toxicity data for selected toxic metals, including aluminum, manganese, and cobalt (Cardwell et al., 1993). The cumulative frequency forms of these distributions were plotted on the same concentration axis for the metal of interest. The degree of overlap between the distribution of exposure and acute or chronic effects concentrations was evident by inspection and, of course, readily calculated if desired. The investigators mapped the location of different species on the toxicity distributions. This offers the additional capability to assess which species are more likely to be affected by overlapping exposure concentrations as well as to contrast the differences in species' acute and chronic responses to each of the metals.

4.1.3. *Regression Models*

Regression analysis involves determining the best fit of sets of observed/measured data to a postulated mathematical model. The models are usually empirical (i.e., their parameters are chosen to provide the best approximation to the data) but express no causal relationships explaining the data. This is not required, however, and some (particularly simple) mechanistic models also can be constructed using regression techniques.

Regression models can be used to establish the best empirical models for estimating single-value parameters. For example, toxicity data on a range of concentrations can be used to estimate the LC_{50} for a species. Furthermore, if a species to be exposed has not been tested, its LC_{50} may be obtained, as a first approximation, from regressions on related species (e.g., taxonomic interpolations). For

example, the LC_{50} value for certain taxonomic groups of organisms has been used to estimate acute or chronic toxicity benchmarks for other species (Suter et al., 1983; Blanck, 1984; Kooijman, 1987; Sloof and Canton, 1983; Sloof et al., 1983; Sloof et al., 1986).

Regression techniques also are useful in deriving empirical models of parameters that vary with time (or with some other independent variable). For example, predictive models of the effects of a stressor on the dynamics of ecological systems require knowledge of the effect of the stressor, in various concentrations, on parameters such as metabolic rate, growth rate, and reproductive success. Experimental or observational data can be used to obtain the best fitting model. Sometimes the dynamic response of entire ecosystems can be reliably predicted by stressor-response functions derived from regression analyses. Well-known examples include the phosphorus-loading models of Vollenweider (1969, 1975, 1976) and Reckhow (1979) in which a few measurements of the physical characteristics of a lake are used to predict the relationship of phosphorus input and subsequent chlorophyll concentrations (i.e., the level of eutrophication). This technique is also suitable for compounds (Giesy and Graney, 1989).

One of the main advantages of a (correct) regression model is that it predicts with measurable confidence. This is particularly the case for predicting effects at (or near) the mean exposure concentration when predictive power decreases as exposures approach the lower or upper limits of the exposure concentrations. The goodness of fit of the model to the data is a function of both. Assuming linearity in the data simplifies the model. The validity of this assumption is best determined by visual examination of a scatterplot of the relationship; the human eye is a sensitive detector of departures from linearity and nonhomogeneity of variance. Often some transformation of the data may be required (e.g., log-log, semilog) before a linear regression model can be postulated. If a workable transformation cannot be found, a polynomial model of an appropriate degree higher than one must be used. In the absence of knowledge about the causal relationships between the variables, caution must be used in adding terms to the polynomial model simply to further improve the statistical fit, lest spurious relationships be incorporated.

4.1.4. *Fuzzy Sets and Fuzzy Arithmetic*

Fuzzy set theory and fuzzy arithmetic offer a different perspective concerning variability and uncertainty from parametric distribution functions (Zadeh, 1965). One potential problem with using parametric statistics to propagate uncertainty in ecological risk assessment lies in defining distributions of uncertain parameters. Uncertainty might take the form of bias and imprecision that result from

inadequate sampling or quantitative measure of the endpoint of interest. These kinds of uncertainty lend themselves to statistical characterization using parametric distributions. Uncertainty might also result, however, from an incomplete understanding of the ecological phenomena; that is, uncertainty that cannot be reduced by merely increasing sampling intensity. It is possible to represent this kind of uncertainty using parametric distributions. For example, using best-guess values for maximum, minimum, and central tendency, triangular distributions can be defined. Similarly, knowledge or an estimation of a range of possible values can be used to define a uniform distribution. Other subjectively defined distributions are certainly possible, such as truncated normal or lognormal distributions.

Assuming the distributed values discussed above are parameters for some mathematical model, Monte Carlo methods can then be used to examine the implications of overall uncertainty in estimating risks. A potential pitfall emerges when the subjective origins of parameter distributions are forgotten when estimating the remaining model parameters, performing the simulations, and interpreting the results. Statistical credence might be incorrectly assigned to model results that are determined by subjective probability that has taken the form of parametric distributions. Another problem might result from the mechanics of Monte Carlo simulations. This approach has an overall "averaging" tendency, and the probability of selecting combinations of parameter values that produce extreme model results varies inversely with the number of model parameters. Even stratified sampling methods (e.g., Latin hypercube) do not guarantee sampling of combinations of extreme parameter values. Thus the Monte Carlo method is not an effective choice for exploring potential extreme model calculations.

In fuzzy set theory, membership functions replace the probability density functions used in statistical error propagation. A membership function consists of a set of values each of which is a "member" according to a certain level of confidence or certainty. For example, an ecological assemblage might be characterized as diverse in its species composition using the descriptors low, medium, and high. Where one membership function (or set of numbers) ends and another begins is known only with some level of confidence, hence "fuzzy sets." The boundaries of these three descriptors might overlap, depending on who assigns the values to any particular assemblage. Membership functions can be used to more effectively represent subjective uncertainties associated with these kinds of model parameters (Ferson and Kuhn, 1992). Fuzzy sets are particularly amenable to incorporating subjective uncertainties into risk estimation. These functions provide an alternative to representing subjectivity by forcing expert opinion to fit parametric distributions.

Manipulation of membership functions, or fuzzy sets, involves the use of a subset of interval mathematics. Interval calculations were developed in the 1950s to examine error propagation in largely engineering applications. Interval calculations use the possible combinations of parameter values that are each represented as a range. For example, if in a simple additive model, parameter A is the interval (2,4) and parameter B is (6,8), the model result lies on the interval (8,12). An undesirable by-product of interval calculations is the continued amplification of model error with the addition of each new and uncertain model parameter. Fuzzy arithmetic was developed to remedy this escalation in model error. In fuzzy arithmetic, the operations of addition, subtraction, multiplication, and division are defined. Fuzzy methods will preserve the extreme combinations of model parameters and the results of fuzzy methods will bound the results of Monte Carlo simulations for the same model, assuming that the parametric distributions used in the Monte Carlo simulations were mapped correctly into their corresponding fuzzy sets.

It is possible to use a fuzzy approach in estimating ecological risks. For example, fuzzy quotients can be used in screening calculations if the exposure concentrations or effects concentrations are not sufficiently known to employ parametric statistics. Similarly, the comparison of joint frequency distributions can be conducted using fuzzy methods by substituting the corresponding membership functions for the distributions of expected environmental exposure concentrations (i.e., acute and chronic toxicity concentrations). Fuzzy calculations for more complex models can be performed using readily available software tools (e.g., FuzziCalc, RiskCalc).

The results of fuzzy calculations can be manipulated to produce analogs to risks, where the probabilities (i.e., [0.1]) are replaced by confidence values associated with possible levels of effects. These fuzzy confidence levels also are enumerated over a range of 0 to 1. Fuzzy arithmetic also can be used to construct analogs to parametric cumulative distributions. These cumulative confidence functions can be used analogously to associate a subjective expectation that a specified level of effect would be realized. Thus the results of application of fuzzy methods in risk integration are somewhat analogous to the methods based in normal statistics discussed elsewhere in this chapter. However, it should be emphasized that fuzzy methods are not homomorphic to parametric methods. Comparisons of results produced by fuzzy methods with statistical methods should not be forced. Fuzzy sets and fuzzy arithmetic represent another way of conceptualizing uncertainty and calculating its impact on model performance. Value judgments concerning the better of the methods (i.e., Monte Carlo, fuzzy arithmetic) should be avoided; the methods are simply different ways of thinking about and examining uncertainty.

Another method that should be explored for risk characterization is the use of a combination of fuzzy and parametric methods to provide a hybrid mathematical approach for estimating risks (Ferson and Kuhn, 1992). Consider a process-oriented model (e.g., population, ecosystem model) where some input parameters are described by parametric distributions and others are represented as membership functions. This hybrid approach might provide the best (least biased) tool to combine the subjective and nonsubjective uncertainties commonly encountered in most risk assessments.

4.1.5. *Extreme Event Analysis*

The analysis of extreme events focuses on estimating the occurrence of low-probability events that are of major (i.e., catastrophic) consequence (Asbeck and Haimes, 1984; Karlson and Haimes, 1988; Mitsiopoulos and Haimes, 1989). Extreme ecological events might include, for example, the probability of local extinction of a species, the successful invasion of an exotic species, pest outbreaks, or the complete functional breakdown of an ecosystem. The very nature of these kinds of events argues for developing and implementing methods for characterizing risks of extreme events that can be added to the arsenal of risk estimation tools.

Normal statistics focus on the mean of the statistical population. Thus statistics based on assumptions of normality might not provide the most effective means for estimating low-probability events. Also, the use of parametric methods is not the best approach for estimating extreme events; that is, 95 percent (or 99 percent) confidence estimates are not necessarily good estimates of extreme events because such estimates are really quantitative statements about the mean. One simple example lies in the nature of the standard normal distribution, where the tails of the distribution must be truncated (at least to non-negative values) to make sense in most ecological applications: Detailed analysis of the tails of this distribution using conventional statistics that focus on the expected value may provide inaccurate estimates of low-probability events. Moreover, by definition, the less likely the event, the fewer the data points that will be available to accurately characterize these regions of the distribution. Although prediction intervals come closer to addressing extreme events, they also are highly influenced by the data that define conditional expectations to the statistics of extremes (e.g., Gumbel, 1958).

Low-probability events can be more accurately assessed by reforming the underlying model to focus on those ecological and stressor circumstances that increase their likelihood of occurring. For example, organisms already subject to stress (e.g., drought, toxics, habitat loss) might be placed at even greater risk by the next stressor, even though the magnitude of the additional stress would by

itself not be expected to produce drastic consequences. Conditions that lead to extreme values of the stressors might be similarly examined and anticipated. For example, combinations of circumstances and mechanical failures leading to severe accidents (e.g., Chernobyl) and spills (e.g., Exxon Valdez) can be characterized, or management activities that set the stage for catastrophic events can be delineated (e.g., forest management practices that result in the accumulation of fuel for major fires, as in Yellowstone, or set the stage for rapid and extensive proliferation of pests once an outbreak occurs). For these kinds of events, the probability density function that describes risk can be constructed using different methods (e.g., statistical, simulation modeling) that are described in this chapter. This function can be partitioned into several conditional risk subfunctions (Karlson and Haimes, 1988): one for high-frequency, low-consequence events; another for intermediate range events; and a third for extreme and catastrophic events.

The statistics of extremes demonstrate that the largest (or smallest) values of a distribution of random samples also are random variables with probability distributions that can be derived from the initial distribution (Gumbel, 1958). The partitioned multiobjective risk method (PMRM) can be used to estimate the parameters that define the distributions of the extreme events (Karlson and Haimes, 1988; Mitsiopoulos and Haimes, 1989). The PMRM provides a tool for a more comprehensive characterization of risk; however, the method is sensitive to the choice of the initial distribution. In many applications, the distribution best representing the fundamental nature of the problem is not evident. This limitation can be addressed, however, by examining the implications of different underlying distributions for the conditional risks of extreme events (Karlson and Haimes, 1988).

Ferson et al. (1989) used an age-structured, density-dependent demographic model to examine the likelihood of quasi-extinction of local populations of brook trout. Demographic parameters were derived from population data summarized in McFadden et al. (1967). By using a Monte Carlo approach, uncertainties assigned to model parameters resulted in a set of possible population projections. These projections defined a function that described the probability of realizing any level of change in the abundance of this trout population, including local extinction. Ferson et al. (1989) discussed the importance of focusing on model processes and regions of the parameter space that correspond with extreme model behavior, instead of attempting to derive the probability of extreme events using statistics that focus on expected values.

4.1.6. *Advantages and Disadvantages*

Empirical approaches to risk characterization offer several clear advantages: (1) well- established quantitative relationships between stress and response permit efficient estimates of risk with associated levels of statistical confidence; (2) the risks are typically easily calculated, given the empirical model; (3) new data can be readily incorporated to refine, extend, or reject the model; and (4) the domain of applicability of the model is often quantifiable. For example, the bioaccumulation of hydrophobic organic chemicals can be predicted using regression models based on the octanol-water partitioning coefficients. The regression holds for chemicals with log K_{ow} less than 4.5 (Mackay and Paterson, 1992). For less-soluble organic compounds (i.e., log K_{ow} greater than 4.5), this relationship provides less-accurate estimates of bioaccumulation because food web effects and other factors become increasingly important.

The main disadvantage of the empirical model approach lies in the data requirements and statistical nature of the relationship between exposure and response. For instance, the quality of the model is strictly data dependent (i.e., no data on system response, no model of this type). The parameters of the empirical model offer no process-level understanding of the relationship between exposure and response. Their values are chosen solely to improve the goodness of fit of the model. Thus the validity of applying the model to new situations is constrained by the ecological and toxicological particulars of the data used to derive the original model and is always problematic. The addition of each new data point directly changes the behavior of state variables in the model.

A major constraint regarding this approach is that for some disturbances it simply may be impossible to construct such empirical models because the entire purpose of the approach is to avoid occasions of the disturbance that could provide the necessary data (e.g., toxic spills, introduction of genetically engineered microorganisms, nuclear winter, ionizing radiation).

Because statistical or empirical relationships imply nothing regarding underlying cause-and-effect relationships, risks estimated empirically require faith in the statistical adequacy of the model as well as the data, and extrapolation of the predictions outside of the data set is not justifiable. This also means that the parameters and coefficients in the models (e.g., the betas in a regression equation) may have little or no meaningful interpretative value that could contribute to understanding the relationship or developing and evaluating mitigation or remediation alternatives. Additionally, ecological and toxicological data commonly make for multidimensional and sparse data sets. Subsequent derivations of empirical relationships without some process-level understanding can lead to statistical "fishing expeditions," where the constraints and assumptions underlying the statistical method of choice are

relaxed or ignored. Finally, in some instances (e.g., multiple nonlinear regression methods), the estimation of model parameters is nontrivial. Final values can be heavily influenced by initial estimates, the complex geometry of the solution space, local minima, and the nature of the estimation algorithm.

Where possible and appropriate, however, empirical models offer good predictive ability at a relatively modest cost in terms of data required. If sufficient data are available, empirical approaches to risk characterization may be the method of choice. Indeed, if risks can be directly estimated to the desired accuracy and precision using existing data, why collect more data or build additional models?

4.2. Process (Mechanistic) Models

Process models are an attempt to mathematically represent the physical, chemical, and biological processes that determine the dynamics of ecological systems and to formulate the toxicological processes that translate stress into response. They are quite literally a priori sets of hypotheses about the causal mechanisms operating in the system the model represents. As such, the inductive step from observation/experimentation on the system to the computer program is followed by computer-generated deductive predictions that are subject to testing to decide on the value of the model.

Process models will become increasingly important in characterizing ecological risks; the sheer number of assessments will require increased use of models. In some instances, avoiding the "experimental" evaluation of the disturbance is the whole point, and models represent the only alternative. Process models continue to be developed to address diverse ecological issues at many levels of measurement. Although interesting and important progress continues in modeling the risk-related dynamics of biochemical systems (Andersen et al., 1987; Gerlowski and Jain, 1983), this chapter focuses on models of at least whole organisms.

The concept of "levels of organization" has been used in an overly simplistic manner in the development of methods for ecological risk assessment. A tidy, nested, spatial hierarchy has been developed to dissect natural complexity into regions or landscapes that encompass watersheds that contain ecosystems; ecosystems consist of communities that in turn comprise populations of individual organisms. In constructing this categorization of complexity, the essential conceptual contributions to ecology associated with these terms often are glossed over or missed altogether. These terms have become convenient ecological shorthand, instead of stimulating conceptually powerful alternative approaches to the description, study, and understanding of nature. We attempt, in this section, to

present these alternative modeling views in a manner that emphasizes their commonality as alternative tools to fashion ecological risk assessments.

4.2.1. *Models of Individuals*

Individual-based models (IBMs) have been developed to describe the growth of individual organisms (Huston et al., 1989). These models are attractive because the primary-state variable is conveniently scaled to the observer; organisms represent tangible entities for mathematical description. Different approaches to modeling individuals have been offered. Some IBMs simply address growth (e.g., FORET; Shugart and West, 1977). Others attempt to simulate behavior, life history, growth, and reproduction. Hallam et al. (1990) developed an IBM that evaluates risks posed to *Daphnia* populations by hydrophobic organic chemicals.

A major advance offered by the individual modeling approach lies in the ability to include aspects of individual biology and ecology that are best described using different units and measures. These growth algorithms can include bioenergetics of growth as well as behavioral attributes and life history phenomena that are not readily formulated in a single differential equation that requires a single conserved unit of measurement (e.g., kcals, carbon, or nitrogen).

Another promise of IBMs lies in their ability to examine the implications of different modes of disturbance on the growth and survival of the individual. By simulating a large number of individuals, IBMs can be used to examine some population impacts of disturbance.

The primary limitations in the development of IBMs are (1) determining what constitutes "individuality" in the basic formalism of the model and (2) the relative availability of data that truly measure the characteristics and attributes of individuals (Bartell and Brenkert, 1991). If the same governing equations are used to describe both the growth dynamics of an individual and the population, it should not be surprising that the aggregate behavior of the individuals closely resembles that of the population. Such similarity, however, is not an emergent property of the IBM.

Individuality might take the form of correlations among values of growth parameters; that is, individuals characterized by higher growth rates might have higher feeding rates, lower respiration (or maintenance) rates, and higher assimilation efficiencies. Individuals that grow more slowly might have the opposite growth characteristics. Whether the relationships are continuous or discontinuous is another problem that must be surmounted in such models and, if discontinuities exist, mechanisms to simulate the times and levels at which switching occurs must be incorporated into the model. The main limitation will always be in corroborating these kinds of assumptions with measurements made

on actual individual organisms. Certainly this approach might be more applicable for organisms scaled similarly to humans (e.g., large body size, longevity, and ability to recapture and remeasure).

4.2.2. *Population Models*

In risk estimation, a population has come to mean a collection of individuals belonging to the same species and occupying the same local habitat. More value could be added to population-level characterizations by defining the population more specifically. The ecologically relevant population may be different from a localized statistical population. Ecological risks posed to a population defined in terms of its genetics (i.e., reproductive individuals, adaptability) may be quite different from risks posed to the collective population. The same may be true for different age (size) classes or life-history stages of a population. Moreover, a local (or statistical) collection may be the wrong population for assessing risks of reduction if the species is broadly distributed throughout the region and is relatively mobile. Thus, risks to the ecologically important population could be over- or underestimated. The lesson for risk characterization is that the population must be defined rigorously and meaningfully if population ecology is to contribute to its full potential in ecological risk estimation. This rigor also will be important in the evaluation and recovery components of an overall risk assessment.

Ecologists have been modeling the dynamics of populations since the early decades of this century (Hutchinson [1978] provides some fascinating historical insights despite the lack of the word *model* in the index). Papers too numerous to adequately review here have resulted from the efforts of ecologists to quantify basic population dynamics (see May, 1973). Two primary approaches to describing population dynamics have been used, one based on changes in the number of individuals (Lotka, 1924), and another based on conserved mass/energy (e.g., Richman, 1958; Slobodkin, 1961; Kitchell et al., 1974, 1977). Steele (1974) combined both approaches in his model of marine zooplankton population dynamics. Models of changes in numbers of individuals cannot be joined to produce multicompartmental systems models representing conserved mass/energy flow. Energy/element-based population models can be easily converted in this manner, and one can recover numbers of individuals from their dynamics.

Effective application of population modeling per se in ecological risk estimation has been limited mainly by difficulties in deriving disturbance-response functions that can be used to alter model parameter values in relation to the degree of disturbance. Despite this limitation, population models

have been used to characterize ecological risks in selected applications (e.g., Barnthouse, 1992; Ferson et al., 1991).

Population models of some degree of species or life-history-stage aggregation form the basis for many, if not all, models of larger-scale ecological systems. Therefore, an adequate knowledge of the range of structural complexity and associated dynamical behavior of population models is essential for constructing models of the larger-scale systems.

An attraction of population models lies in their relative structural and mathematical simplicity. Population dynamics is characterized as a constrained compound interest problem. The complexity of nature is aggregated into the estimates of the model parameters (e.g., maximum growth or population ingestion rate, resource satiation levels, and density-related carrying capacity). Analytical solutions of time-varying population size can be obtained for many of these models. Matrix manipulation or numerical integration methods are routinely used to solve some of the more-complex, multiple-age (size) class population models. Models of competing populations or predator-prey populations have been thoroughly studied (May, 1981).

Another advantage of the age (size) class demographic population models lies in the ability to derive parameter values from repeated observations or measurements of the population. Detailed process-level understanding is not required to assemble a model that can extrapolate current transition probabilities or growth rates into estimates of future population size. This same lack of understanding of the underlying mechanisms regulating population dynamics also can be a disadvantage; for example, the model fails to alert the user to circumstances that change the demographic model parameter values (i.e., the effects of a stressor).

Certain demographic model formulations are inherently unstable, converging to zero or diverging to infinity unless the parameter values are balanced to greater precision than could ever be measured for natural populations. Density-dependent formulations must be posited for numerical stability. It remains nearly impossible to empirically demonstrate the existence of such mechanisms in real-world populations, although significant efforts continue to be directed at establishing plausible density dependencies (e.g., compensatory effects in fish [Rose et al., in prep.]).

Population models also can exhibit chaotic behavior in certain regions of their parameter space (May and Oster, 1976; Schaffer, 1985). Chaotic models produce aperiodic results despite an underlying mathematical determinism. This model behavior appears as more of a mathematical artifact of the basic model construct than as a characteristic of natural populations (O'Neill et al.,

1982b). The potential for chaotic behavior must be kept in mind when these kinds of models are used to project population sizes over time.

The use of such models to estimate ecological risks will undoubtedly continue. As with process-level models, model verification, validation, and analysis will remain important issues to be addressed in using population models to characterize ecological risks.

The bioenergetics approach to population modeling is based on a mathematical description of the energy budget of the population (Kitchell et al., 1974). Growth is modeled as the integration of rates of energy input minus energy losses for the population. The bioenergetics population models have been used to (1) project future population size given estimates of the bioenergetics of growth, food availability, and environmental conditions (e.g., Kitchell et al., 1974, 1977); and (2) to estimate feeding inputs by fitting the model to measured growth data (Rice and Cochran, 1984).

The bioenergetics equations used in these models permit the incorporation of stressor effects, particularly sublethal effects, as alterations in the energy budget of the organisms that make up the population (e.g., Bartell, 1990). The equations describe the physiology of growth, expressed as the processing of energy (or biomass expressed as carbon). Stressor effects can be represented as changes in the rates of the physiological processes that determine growth in the model population (O'Neill et al., 1982a, 1983; Breck and Bartell, 1988; Bartell et al., 1992). A limited amount of experimental evidence has been collated that supports this approach for a variety of chemical stressors, physiological processes, and organisms (Bartell et al., 1992). The toxicological literature needs to be comprehensively surveyed and analyzed to determine the extent to which this approach usefully applies (see section 4.4.2 for further elaboration of this point).

As with the demographic representations of population dynamics, bioenergetics models have limitations that can impede their application for ecological risk estimation. One limitation is the incomplete quantification of the bioenergetics of growth for species of interest. To implement a model for a particular population, data from several species are used to estimate the values of model parameters. Complete descriptions of the bioenergetics of growth have not been assembled for many species.

The bioenergetics models are typically sensitive to estimates of energy inputs. For example, fish growth as predicted by the bioenergetics models (e.g., Kitchell et al., 1974, 1977; Rice and Cochran, 1984) relies heavily on the estimates of temperature-dependent feeding rates (Bartell et al., 1986a). Values of fish feeding rates under nonlaboratory conditions are nearly impossible to obtain.

Temporal changes in diet composition (and its caloric value) also can introduce uncertainty in estimating relations between consumption and growth.

Another limitation in the bioenergetics models is the description of the population as a unit of biomass, carbon, or its caloric equivalent. Size (age) structure, which characterizes the demographic models by definition, is typically lacking in the bioenergetics models. This level of aggregation may limit the applicability of these models, particularly if specific ages or sizes of organisms are differentially sensitive to the stressor(s) of concern.

One potentially powerful approach to estimating risks to populations lies in the integration of bioenergetics and demographic methods. The process-based bioenergetics models might be used to establish stressor-response relations in terms of physiological understanding. Despite data limitations, separate bioenergetics models might be developed for different age or size classes to provide the necessary detail in population descriptions. The results of the bioenergetics models then could be used to estimate the macroscale parameters required in a demographic description of the overall stressor effects on the growth dynamics of the population.

4.2.3. *Community Models*

In ecology, the concept of community has its roots in the early studies of Clements (1916) and Gleason (1926) on the formation and successional change in plant aggregations. Although these two workers, and later adherents, differed widely on the mechanisms of establishment, change, and ultimate state of communities, they shared an emphasis on biotic interactions. The physical environment, although crucial as a whole in determining the type of community found, was disregarded as a temporally varying force that could shape successional change. The emphasis was placed on the biota, its diversity, and the mechanisms by which species were packed into niches or displaced those already in place. This emphasis was continued in the development of island biogeographic theory (MacArthur and Wilson [1967], who focus on the probabilities of colonization of isolated patches of habitat by species and on their rate of extinction). Thus community models would be expected to focus on processes that reflect the outcomes of the interactions among the biota (e.g., colonization, successional change, biodiversity, niche breadth). Because these interactions are the result of the attributes of individuals and populations, and because in the aggregate they form processes important to the ecosystem and to the landscape, there is quite obviously a very fuzzy line separating models of community from the other category of models. At best, we may categorize models of community as containing interactions of three or more species (to separate them from

individual-based models and two-species population models of competition and predation), with minimal inclusion of temporally varying parameters representing the physical environment (to separate them from models of ecosystems). In addition, we may add the constraint that models of community not explicitly consider two- and three-dimensional spatial heterogeneity (to separate them from models of entire landscapes).

With the caveats above in mind, what is left? The answer must be, from the standpoint of methods to assess ecological risk, not much. Perusal of the pages of ecological journals such as the *American Naturalist*, *Ecology*, or those published by the British Ecological Society yields dozens of models of the community processes enumerated above. These models, however, were designed to propose or assess some theoretical construct of community ecology and were not expected to include algorithms that would permit assessment of risk from some stressor. All of the community properties are properties of ecosystems as well. Furthermore, the nature of most stressors is that either they are transported via some physical component of the system (e.g., air or water) or they directly modify the physical environment. Thus, although specific models of community are vital as tools to develop and modify community theory, the risk assessment model will usually incorporate the community theory in the context of both the biotic community and its physical environment (i.e., in models of the ecosystem or the landscape).

One group of community models, however, offers some potential in terms of risk assessment tools—models of food webs (Cohen et al., 1993; Pimm, 1982), which incorporate many of the attributes of community. For instance, certain impacts of biodiversity are included, as are elements of species' interactions via the trophic pathways. The models can be constructed using graph theory (Pimm, 1982) and are largely qualitative, although certain quantitative variables can be computed (e.g., the mean length of food chains). Because food webs, varying in degree of detail, are documented for a large number of ecological communities (Cohen et al., 1993; Pimm, 1982), such models may serve as a first-approximation qualitative predictor of certain kinds of response to a stressor that can potentially eliminate designated species from a community. Because the predictions are based on qualitative data, however, caution must be used in interpretation.

4.2.4. *Ecosystem Models*

In risk characterization, "ecosystem" too often refers to an entity that corresponds more closely to habitat, as evidenced in the framework (e.g., lake ecosystems, wetland ecosystems, coastal ecosystems). One important contribution of the ecosystem concept to ecological theory lies in the

understanding that biological (e.g., excretion and egestion) and ecological (e.g., predation, decomposition, nutrient cycling) processes can modify the physical, chemical, and biological environment. The role of these feedback mechanisms in determining productivity and maintaining system functional integrity has been long recognized in ecology (e.g., Lindeman, 1942; Hutchinson, 1948). Other key contributions of ecosystem theory include recognition of the scale dependence and hierarchical structure of ecological systems (Allen and Starr, 1982; O'Neill et al., 1986), the potential for discontinuous behavior in system function (e.g., catastrophe [Thom, 1975]) and structure (e.g., strange attractors [Schaffer, 1985]), and relations between energy flow and nutrient cycling as they influence system recovery from response (DeAngelis et al., 1989a). These advances in ecosystem understanding should be considered carefully in the development of ecosystem-level endpoints for risk analysis.

Ecosystem models explicitly address biotic-abiotic constraints, interactions, and feedback. They are by definition multicompartmental models (Wiegert, 1993). Attempts are made to represent relevant physicochemical characteristics of the environment that influence and in turn are influenced by biological interactions (e.g., predator-prey relations, differential resource use). This environmental complexity is aggregated into the parameter estimates and assumptions of the population and community models.

Simulation models of the dynamics of ecosystems are constructed using the relationship—from systems science—that structure of the system (or model) interacts with function to produce behavior of the system (or model) as a whole. Because use of the words structure and function in this context follows the definitions from systems science, and not ecology (Hill and Wiegert, 1980), we briefly reiterate them (figure 9-1).

Structure is the abstract (conceptual) part of the model; the "box and arrow" diagrams illustrate the structural aspects of the model. Setting out the structure of a model is an extremely important step, for it is in this operation that the capabilities and goals of the model are incorporated. Structure determines the degree of complexity of the model and largely determines the number of parameters and thus the informational requirements of the model.

The functional components of the model constitute the specific occupants (e.g., species or aggregated groups, abiotic storages) of the "boxes," plus physical components (e.g., light, temperature, precipitation), and finally the functions and specific parameter values.

Dynamic behavior of the model through time can be measured by changes in standing stock (e.g., number, biomass, energy) and by changes in fluxes between compartments. The bookkeeping

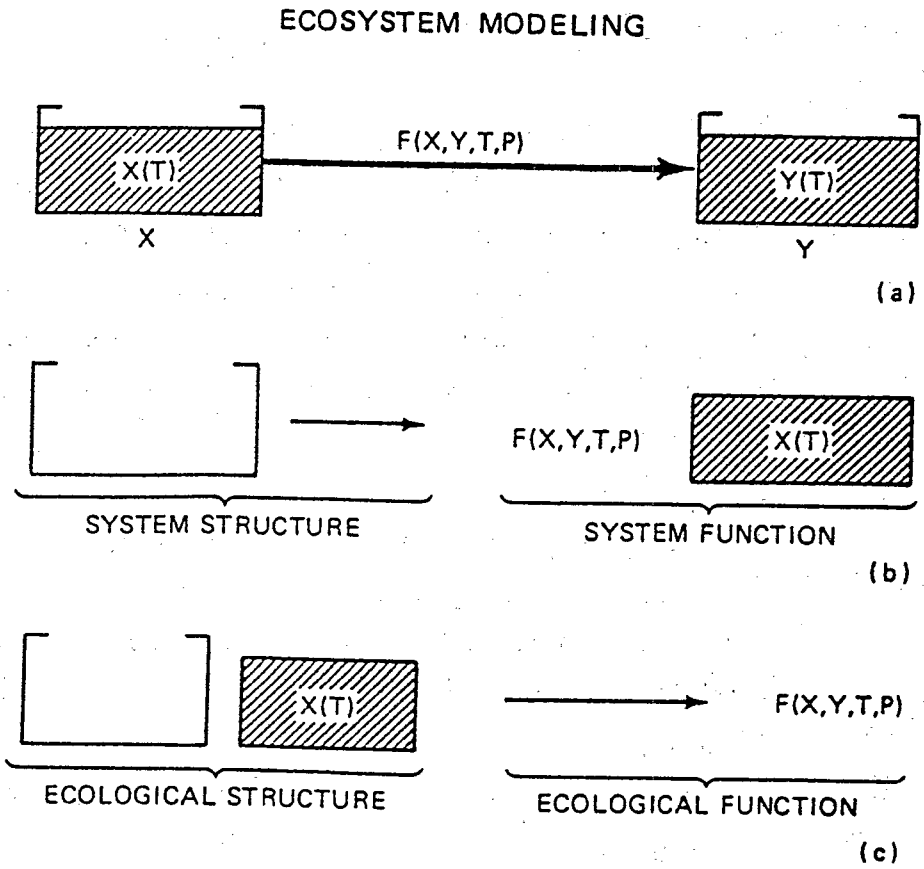

Figure 9-1. Identification of structure and function in a system diagram (redrawn from Hill and Wiegert, 1980).

unit(s) of choice may enter and leave the system from the surroundings. In addition to these sources and sinks, models explicitly dealing with colonization and/or successional change must provide for the entrance and exit of new structural (species) compartments.

The primary advantage of the ecosystem models lies in their ability to incorporate detailed formulation of the processes believed to determine the production dynamics of the system as well as the ability to address physicochemical measures of disturbance (e.g., energy flow, material cycling, habitat alteration).

The potential for additional detail also can be a potential for misuse in applying ecosystem models to risk estimation. The data requirements for model development and implementation can quickly exceed the amount of information commonly available for specific systems. Values for remaining parameters must be estimated using published data that often are not directly applicable to the system of interest. Uncertainties introduced by a complex array of parameter values can produce model results too imprecise to be of use in decision making. Thus, as we emphasized above, the initial structural diagram of the model must be carefully tailored to meet the goals of the model without adding unneeded detail.

Because mechanistic ecosystem models incorporate causal statements into the functional components, they have the disadvantage that their predictive accuracy is completely dependent on the correctness of these causal mechanisms. Thus verification and validation (see sections 4.5.2 and 4.5.3) are vital tests. Once verified and validated, however, such models have the advantage that their predictions are not constrained to an observed range of perturbations on a real ecosystem, as are those of empirical models. Mechanistic ecosystem models also are more costly to construct than empirical models; however (as discussed in section 4.1.6), if data suitable for mechanistic models are not available (or cannot be obtained because of the nature of the stressor), then there are no alternatives.

These models are categorized on the basis of spatial and temporal scales and the degree of heterogeneity they are capable of representing. The temporal scale of the model will be fixed by the goals of the model and the rates at which processes operate in the target ecosystem. Spatial heterogeneity can be categorized in three ways: (1) patch boundaries fixed and within patch-successional change is slow with respect to the temporal scale of the model simulations (e.g., the tidal creek, creek bank, and high marsh habitats of intertidal marshes of the eastern United States [Wiegert, 1986]); (2) patch boundaries fixed—successional change within the patches is part of the simulation goal of the model (e.g., models of forest habitat where successional age determines

suitability as a population source or sink [Pulliam et al., 1992]), or successional change in a marsh in response to environmental degradation [Costanza et al., 1988]); (3) patch boundaries and within patch—succession varying temporally on a small scale with respect to the temporal scale of the simulations (e.g., disturbance models, where patch creation is a function of some outside force acting in a systematic or stochastic manner [see, e.g., Wiegert, 1979]).

Because of the increasing need to model the consequences of stressors applied to the environment, increasingly more process models will be constructed specifically for such applications (e.g., Bartell et al., 1992). Because most ecosystem process models, however, already contain explicit formulations of the processes that are potentially affected by stressors, many can be adapted rather easily to this purpose (e.g., Suter and Bartell, 1992).

4.2.5. *Landscape and Regional Models*

Certain disturbances imply ecological risks at spatial scales commensurate with landscape or regional models. For example, acid deposition, radionuclide transport, and agricultural application of pesticides and fertilizers may portend ecological responses at the scales of headwater watersheds, large river systems (e.g., the Mississippi River basin [Thurman et al., 1992]), or large regions with boundaries determined by prevailing weather (usually wind) patterns (e.g., acid rain or Chernobyl). The methods used to address these larger-scale risks are essentially those of the spatially explicit ecosystem models. The larger size of the spatial scale, however, invites the use of special technologies (e.g., geographic information systems [GIS]) and imposes some constraints.

These landscape and regional models could be considered to be simply larger, more spatially diverse ecosystem models. Although they share a property that is found in a few ecosystem models, it is one that is a requisite in the large system models: the recognition that not only must spatial heterogeneity be modeled, but that the location of each spatially distinct type relative to the others is also important. For example, the landscape watershed model containing agricultural land, floodplain forest, and river must clearly delineate the location of each with respect to the other. The dynamics of such a model will be quite different if all of the floodplain is forested as opposed to the system in which some of the floodplain is used for agriculture.

Spatially explicit models of landscape dynamics that might be used to address larger-scale risks are being developed (e.g., Turner and Gardner, 1991). These models are potentially useful for examining the propagation of disturbances across landscapes (e.g., fire, insect pests). Landscape and

regional models are potentially useful in risk analysis because of the inherent spatial nature of ecological disturbances.

Larger-scale models offer advantages and disadvantages. The major advantage lies in more realistic spatial-temporal description of environmental factors that influence risk. Known locations of point sources can be represented in the model. Spatial and temporal variability in climate, soils, vegetation type, and disturbance can be specified. Information pertinent to estimating risks is not ignored, which is in contrast to the implicit simplifying assumptions of nonspatial or point models. Another advantage specific to these models is the ability to map risk back to time and place in the system of interest. Finally, spatial models might also be used to design and evaluate alternative mitigation or restoration activities.

The tradeoff for these large spatial models concerns the additional data required to develop and execute the model. As with the ecosystem models, a primary drawback in the development and application of landscape and regional models is the number of initial values, parameter values, and boundary values required to implement models of even fairly simple construct (e.g., Bartell and Brenkert, 1991).

Another consideration lies in the magnitude and potential difficulty of the computations required for simulation. Typical model structure is based on a finite element or finite difference representation of system spatial structure; corresponding formulations often are sets of coupled nonlinear partial differential equations. Obtaining accurate, stable, and efficient solutions of these equations remains an active area of science in and of itself. Correspondingly, the number of scientists and institutions with these kinds of capabilities is small in comparison to those able to develop and use empirical approaches or simpler models to estimate ecological risks. Nevertheless, the ability to model the system in exacting detail is a seductive goal.

4.2.6. *Aggregation and Disaggregation*

Because of the need to compromise between model simplicity and the extreme complexity (diversity) of even the simplest of real systems, aggregation of species and life-history stages is necessary in conceptualizing the abstract structure of almost all models of ecosystems, landscapes, and regions. The problem is how to accomplish this aggregation without compromising the goals of the model. To a large extent, experience with the targeted ecological system is invaluable. A group of ecologists familiar with the system (including the modeler, if at all possible) can often solve many of the aggregation problems for the initial model on an intuitive basis, much as the same process is used

to formulate some of the functional relationships for which data might be initially absent. For example, if the goals of the model require risk assessment for specific species of organisms in the system, then the species must be represented as individual-state variables, perhaps even further disaggregated as separate life-history stages. State variables (i.e., compartments) that are known (or suspected) to be vital to the goals of the model or to the dynamics of the system should not be aggregated with variables that have greatly different specific rates of flux. As a trivially obvious example, one should not aggregate detritus-feeding fish and aerobic bacterial decomposers if the dynamics of the resulting aggregated-state variables are important to the goals of the model.

Other problems of aggregation are much more subtle. Wiegert (1975, 1977) studied simple thermal communities with the goal of experimentally testing predictions from various levels of aggregation in the field. Five levels of aggregation of a simple cyanobacterial-ephydrid fly system produced a surprising result. As expected, the best predictor was the most disaggregated model, where most life-history stages of the fly were in separate compartments, and a great deal of complex detail was incorporated in the functional control of the spatial heterogeneity of the cyanobacterial mat and of fly reproduction and growth. Various increases in aggregation were made, the most extreme being reduction to a simple two-compartment model of blue-green algae and the consumer fly. When these were tested against the independently observed behavior of the field system, all intermediate stages of aggregation were found to be worse predictors than the most aggregated case. The explanation lay in the degree to which aggregation (simplification) of the structure of the model was paralleled by simplification of the functional attributes. Complex structural "webs" that were not supported by functional detail produced worse predictions (often instabilities) than the model using extreme aggregation of both.

Too much disaggregation in models results in such a great number of compartments, fluxes, and consequent parameters that falsifying predictions of the model becomes more difficult, or at least understanding which causal mechanisms may be responsible for an erroneous prediction may be difficult.

Similarly, Wlosinski and Minshall (1983) studied the behavior of a model of stream invertebrate production by varying the number of state variables used to describe these invertebrates. At the finest level of resolution, 37 taxonomic populations described the invertebrate community. Other levels of aggregation used 15-, 8-, and 1-state variables to represent invertebrate production dynamics. In comparisons with field measurements, the 8-variable functional description provided the most accurate model results.

In summary, "rules of thumb" about model aggregation/disaggregation are as follows: (1) do not aggregate components with greatly disparate rates of fluxes; (2) do not greatly increase the disaggregation of the structural aspects of the model without a corresponding increase in the sophistication of the functional relationships and controls; and (3) disaggregate models only insofar as required by the goals of the model to facilitate testing.

4.2.7. *Implementation*

Construction and implementation of mechanistic or process models may be highly individualistic and system/stressor oriented. Nonetheless, we can specify two general pathways or strategies: the submodel approach and the approach that begins with a general model and proceeds to more detailed models. This latter approach is often referred to as a process of going from the generic model to the site-specific model.

In the first case, the detailed site-specific model is developed by chaining together a series of models of well-defined (and often relatively loosely coupled) subsystems. As an example, consider the development of process models of plant productivity, the aboveground plant biophage-based food web, the surface and underground plant saprophage-based food web, and the anaerobic microbial decomposer web. Connecting each of these via their appropriate pathways of interaction and transport/transformation yields a detailed system model. This approach is most useful when available information about each subsystem is sufficient to provide comparable submodels.

When the system is less well known, or at least some of the subsystems are data poor, the more common approach is to begin with a simple and general model of the system, incorporating whatever is known and using literature and best estimates for the remainder. Such a model serves to pinpoint not only areas where further data are needed, but simple sensitivity analyses (section 4.5.1) can provide information on the parameters that must be evaluated carefully and those for which estimates are sufficient, at least initially. With continued research on the system, such a model becomes not only more detailed but also more site specific. The time constraints on the risk analysis process may often preclude the development and integration of detailed submodels or the continued refinement of a more general system model. In such cases, the simple general model can be constructed, rendered site specific with whatever data are available, and used.

4.2.8. *Visualization*

At whatever scale they are employed, mechanistic models can quickly become very complex in terms of both numbers of compartments and degree of interaction between compartments. Sensitivity analyses, "what if" scenarios, and other manipulations of such models are at once their strength and weakness. Simple printed lists of output fail to portray the richness of the simulated results, particularly to the targeted users of such models, who may have little or no knowledge of the varied dynamics of the system suspected to be at risk. Thus programs are needed that will take the voluminous output of such models and visually compare and portray the significant differences in simulations. Simple two-dimensional and three-dimensional static graphs are useful, as are sequential graphic displays. In sequential displays, the length of the interval between graph changes will determine if the visual presentation shows a series of disjunct changes or a virtual animation. In all cases, the usual way to proceed is to take the data from one to several different simulation runs and then to develop the visual presentation from the data stored as random access files, which permit almost instantaneous retrieval.

4.2.9. *Advantages and Disadvantages*

In contrast to empirical models, mechanistic models include statements of causation. Indeed, they are organized and coupled hypotheses explaining how the abstract structure of the model system interacts with the functional attributes of the specific model to generate behavior of the whole. As such, these models have the advantage of incorporating information about system behavior that does not depend on prior observation of the system under the influence of a stressor. This generality comes at the cost of a much greater knowledge of the ecology of the system components than that required by empirical models and with a much greater level of uncertainty attached to predictions relative to regression and other empirical models. In many risk assessment situations, however, an empirical database on the effects of a stressor is not available. In such cases, when the threatened system is large or greatly valued and the threatened effect of the stressor is large or long lasting, mechanistic models often are the only useful tools for assessing risk.

4.3. Physical and Experimental Models

Artificial ecological systems can be used to measure or quantify the response of ecological systems to disturbance or stress. In these systems, the stressor is physically applied to a representative ecological receptor system. Impacts (or endpoints) can be measured directly, and the

degree of experimental replication provides information for quantifying uncertainty and estimating risk. Thus artificial ecological systems may be powerful instruments for risk integration. Additionally, one way of testing predictions of both empirical and mechanistic models is by using physical analogs of the real system (cosms) or a particular expression (usually small) of the real system itself (field-scale experiments). Cosms of various size permit various degrees of replication, depending on their size. The difference between the cosm and the field-scale experiment is, however, more than one of size and the degree to which movement across the boundaries of the system can be controlled.

4.3.1. *Cosms*

Although the entire field of microcosmology cannot be exhaustively treated within the scope of this chapter, several observations, important points, and recommendations regarding cosm use in assessing ecological risks must be noted for the purposes of this discussion. Artificial ecosystems of various size and construction have been used for decades to study the ecological impacts of temperature alterations, nutrient additions (e.g., Conover and Francis, 1973), or chemical contaminants (Giesy, 1980). Cosms used to assess the ecological impacts of anthropogenic stress have included simple laboratory aquaria, larger and more elaborate marine tanks, enclosed portions of freshwater and marine systems, limnocorrals, littoral zone enclosures, soil columns, old-field enclosures, artificial streams, forest canopy enclosures, variously scaled growth chambers, and many others. Certain disturbances of importance in ecological risk assessments lend themselves to study in artificial or experimental ecosystems (Suter and Bartell, 1992).

Ecological scaling is perhaps the most important issue in designing, constructing, and using cosms to perform basic ecological research or to estimate ecological risks. Historically, such scale considerations were determined by the dimensions of experimental units that were readily available from biological supply houses, pet shops, and outlets for construction materials, as well as by the capabilities of investigators or craftsmen in the basic trades. Not surprisingly, most aquatic cosm work has been scaled by the availability of laboratory glassware and commercial aquaria. By scale is meant not only the physical dimensions of the cosm, but also the nature of boundary conditions and the biological components of the system, particularly in terms of initial conditions and sampling. Although the physical dimensions and boundary conditions largely dictate the fundamental performance of cosms, light, temperature, atmosphere, soils, or water included as part of the design determine the habitat suitability of the system. Physical size influences the transmission of conditions

at the cosm boundaries (e.g., light, temperature, humidity, nutrients) to internal environmental conditions and determines what kinds of organisms can be included in the experimental design. The fact that organisms cannot be miniaturized places the ultimate limits on using microcosm studies for assessing risk or conducting research.

Scale also constrains the initial conditions in cosm studies. The number or biomass of different kinds of organisms added to cosms is determined by the carrying capacity of the microcosm referenced to the duration of the study. Knowledge of the life-history characteristics and basic energetics can be used to estimate the size of initial additions of primary producers, consumers, and higher trophic level representatives. The objective is to avoid distorting ecological interactions (e.g., grazing, predation, resource competition) that can bias cosm results and decrease the value of the resulting data for extrapolating to risks posed by the stressor in the larger ecological systems of interest. One approach to defining initial conditions is careful estimation of the carrying capacity of the cosm biota in relation to energy and resource availability. Another approach is simply to inoculate with comparatively small numbers of the constituent organisms and let the cosms take their course toward some quasi-equilibrium state before beginning the experiment or imposing the stressor. The first approach is perhaps more defensible in theory, but the second approach is more often used.

Scale also influences the sampling of cosms to measure the impacts of stressors. Clearly, the total size (volume) of the system determines sample size. Sampling should not pose greater disturbance on the cosm than the stressor being studied; indeed, sampling should be designed to impose minimal disturbance. Sample frequency must be similarly planned in relation to the characteristic frequency of the ecological endpoint, within constraints imposed by cosm dimensions. Sampling plans also should address replacement of sample volume with fresh media or substrate. In some cases, it might prove necessary to destructively sample the entire cosm, with the number of replicate cosms that must be constructed initially added to overhead.

Using cosms to estimate ecological risks poses an additional demand not necessarily encountered by basic cosm researchers. If we insist that risk connotes a probability, the cosm experiments must be designed to characterize not just an ecological effect, but to quantify the probability of the effect. The increased number of replicates required to reliably estimate such probabilities might severely constrain the use of cosms (and other physical models) for characterizing ecological risks.

4.3.2. *Field-Scale Experiments*

Whole-system manipulations have contributed to the basic ecological understanding of ecosystem response to disturbance (e.g., Schindler et al., 1985; Carpenter et al., 1985). The major advantage of this approach is that real-world systems are the object of study; realistic levels of complexity are represented by definition. Thus the results of manipulative experiments do not have to be extrapolated to the real world, as in the case of manufactured ecosystems (i.e., cosms). Nonetheless, extrapolation problems still remain if the systems are considered representative of similar systems at other locations.

The major drawbacks of whole-system experiments include (1) difficulty in replication with attendant high variance and low statistical power; (2) expense in manipulation, sampling, and data processing; (3) duration of the experiments; and (4) uncontrolled conditions that can confound interpretation of results.

Difficulties in replication attest to the few places where whole-system experiments can be performed. Experimental sites currently are limited to several U.S. Department of Energy National Environmental Research Parks (NERPs), several National Science Foundation-sponsored Long Term Ecological Research sites (LTERs), and a few EPA Centers of Excellence. Even at these locations, access and permission to perform manipulative experiments on whole systems are justifiably restricted and tightly regulated. In the case of toxic chemicals, whole-system manipulations involving the introduction of toxicant require expensive waste treatment. Many such experiments are strictly regulated or prohibited (e.g., field studies using genetically engineered organisms, studies using radioactive materials as tracers).

Such experiments may involve manipulation of any combination(s) of abiotic components, autotrophs, and heterotrophs. In the case of abiotic components, such as organic and inorganic nutrients, additive experiments are possible but depletions are not. Such additions are often easier in aquatic systems and human-managed (i.e., agricultural) systems. More commonly, biotic components are experimentally manipulated by removal experiments employing undisturbed controls rather than by additions, which are usually not possible because of the sheer magnitude of the effort required. In either case, whole-system manipulation, the associated sampling effort, and sample processing combine to challenge even the most generous of project budgets. Where whole-system experiments are under consideration for assessing ecological risks, a cost-benefit analysis comparing the expected value of the research versus the costs of restoring ecological systems damaged by the stressor of concern is well advised. Inadvertent large-scale addition "experiments" can result from the hasty,

widespread commercial introduction of a toxic compound or the accidental or deliberate introduction of an organism that turns into a pest. Much has been learned in the past from the results of such damaging perturbations. Restoration, remediation, and containment might prove more economical than whole-system manipulation, particularly if each event is treated as an "unplanned" whole-system experiment with properly designed follow-up monitoring and analysis.

The anticipated duration of field experiments also determines the efficacy of this approach to risk integration. Whole-system experiments might be profitably applied in systems with relatively short turnover time in relation to the funding duration of most sponsoring organizations (e.g., 1 to 3 years). Certainly, the longer the turnover time for the system of interest, the fewer the experiments that can be performed and the greater the associated decrease in statistical power. Nevertheless, the spatial-temporal scale of the stressor might provide the real constraints on designing and executing a whole-system manipulation. For example, the rate of degradation of a chemical stressor of concern could determine the duration of any field experiment designed to assess its ecological risk, since some chemicals of concern are long-lived in the environment (e.g., metals, organochlorine pesticides, polynuclear aromatic hydrocarbons [PAHs], polychlorinated biphenyls [PCBs], many radionuclides).

The utility of whole-system experiments also is influenced by uncontrollable factors that might affect the results of the study. For example, experience suggests that field studies are seldom performed under the long-term average weather conditions, particularly in highly seasonal environments. Each year seems a meteorological anomaly (routinely used to explain the unexpected results of the study). Additionally, larger-scale systems may be experiencing several stressors simultaneously, with subsequent confounding effects on interpreting the results of any manipulation. For example, the Great Smoky Mountains National Park is currently experiencing the impacts of the baslam wooly aphid, atmospheric acid deposition, and increasing populations of wild boar, with the gypsy moth looming on the horizon; additional whole-system experiments seem to portend insurmountable confounding factors in the Smokies. Similarly, human-dominated systems (and most are) will prove challenging for designing manipulative experiments that promise clearly interpretable results.

The limitations of whole-system experiments outlined in this section might seem to remove them from serious consideration for assessing and integrating ecological risks. Importantly, however, these systems exhibit exactly the kind of complexity that is relevant to assessing ecological risks in the world where risks matter. Therefore, it is imperative that striving for opportunities to measure

ecological impacts in these complex systems remains an objective in developing capabilities in ecological risk assessment.

4.3.3. *Advantages and Disadvantages*

Both cosms and field-scale experiments have an inherent advantage over empirical and mechanistic models in that they involve associations of living organisms representing a concrete ecosystem, no matter how artificially contrived or simplified. Cosms, in particular, have a long history of use in screening for the toxic effects of stressors on reproducing populations. Field-scale experiments, when conducted on representative samples of the system at risk and then replicated, can provide data from which empirical models can be constructed. Both cosms and field experiments, however, are expensive compared with simulation models (and regression models where data permit). Moreover, they are often only marginally representative of the system at risk, and questions of scale, particularly with cosms, often render data from the experiment problematic when used to assess risk.

4.4. Example Applications

Ecological models, both empirical and mechanistic, have been used in many ways to predict the effects of acute and chronic perturbations. The scope of this chapter does not permit an in-depth discussion of the range of these applications and, in any case, the chapter is not intended to be a literature review. Instead, we focus on three examples to give some idea of the broad scope of model type and usage. One of the models is largely an empirical phosphorus- loading model for lakes, the second combines a population toxicity simulator with a mechanistic pelagic ecosystem model to assess risk to fish, and the third uses a mechanistic estuarine model to predict the effect of manipulating a population.

4.4.1. *Phosphorus-Loading Models*

Vollenweider (1969) used observed empirical relationships between nutrient availability and depth, together with outflow losses and sedimentation, to develop an empirical model of phosphorus-loading that would be adequate to predict levels of eutrophication in unstratified lakes for which only a few required physical measurements were available. Later improvements in the model incorporated additional attributes, such as lake basin morphometry, water residence times, and phosphorus residence times (Vollenweider, 1975, 1976), that permitted the application of the model to

a wider class of lakes. Modifications to adapt such models to stratified lakes or rapidly flushed reservoirs require only that adequate databases on such lake types are available.

The conditions for the success of the predictions made with such models depend almost wholly on the adequacy of the database used for fitting and evaluating and the skill with which the model fitting is done. The drawback, apart from the necessity of having a large existing data bank, is that such models are not justifiably extrapolated beyond the range of the data with which they were constructed. By contrast, mechanistic models are valid, at least in theory, over a wide range of behavior by the variables of state, since the data-fitting is done at the parameter level rather than at the level of the behavior of variables of state.

4.4.2. *Toxicity Risk Models in the Ecosystem Context*

Bartell (1990) explored the feasibility of predicting the effects of toxic chemical stressors on the fish populations of the pelagic zone. Following the methodological schema of O'Neill et al. (1982a), Bartell employed a population-specific toxic effects model together with a pelagic ecosystem model (see section 4.2.2). The former was used to generate a toxic effects matrix from the relation between stressor concentration and population growth rate. The effects matrix was then coupled to a mechanistic pelagic ecosystem model. In this manner the effects of the toxic stressor on populations could be evaluated in the context of the entire ecosystem (e.g., Bartell et al., 1988).

The conditions for the adequacy of such models were established in sections 4.2.2 and 4.2.4. Essentially, the conditions embody the uncertainties involved in fitting the toxicity data to the population growth model and the verification of the ecosystem model to which it is connected.

4.4.3. *Ecosystem Models as Population Predictors*

When the data for an empirical model are not available or cannot be collected (see section 4.2.4), mechanistic predictors are the only recourse. In Spain's Ria Arosa, the probable effect of increasing the number of mussel rafts on the mussel harvest per raft was desired. Unfortunately, only one Ria had been exploited heavily in this manner. Thus no data existed on the response of the variables of state (productivities) to various levels of stocking. Furthermore, the consequence of increasing the number of rafts, with the consequent movement of families into the region and a possible decline in the mussel harvest per raft, was a socially unacceptable risk. Although this example deals with the management of an economic resource, it is illustrative of the conditions under which a mechanistic model might be used in risk assessment.

In collaboration with a large number of Spanish and North American scientists, a detailed mechanistic model of the estuary and the mussel rafts was begun (Wiegert and Penas-Lado, 1982), and increasingly more detailed modifications of the model have been made during the ensuing decade (Wiegert and Penas-Lado, in prep.). The important prediction from this model was that increasing the number of commercial raft-borne mussels in the pelagic zone of the Ria Arosa by 50 percent would increase slightly the overall harvest of mussels but would significantly lower the production of mussels per raft and thus lower the income of the individual families that owned the rafts.

4.5. Uncertainties

Uncertainties in using purely mechanistic models for prediction stem entirely from the uncertainty involving the degree to which the structure of the model is correct and the functional controls and parameter values are real. In the present case (example 4.4.3), the model was verified (section 4.5.2) against the data from the situation in the 1981 to 1986 period collected by a team of Spanish and American ecologists. During this period the estuary supported approximately 2,000 rafts, and the model predictions were verified against the productivity and standing crop data. Evaluation (validation, section 4.5.3) could only come about if the rafts were to be increased (or decreased). Since rafts are family-run, such changes in raft numbers could be accompanied by major perturbations in the social and economic structure of the region. Even when verified but not validated, however, such predictions from mechanistic simulation models are still useful as cautionary tales suggesting certain lines of research that may yield ever better prediction. (Note that an alternative test, involving larger rather than more rafts, appears to be taking place and may yet yield validation data for this model.)

4.5.1. *Model Sensitivity Analysis*

Sensitivity analysis measures the partial derivatives of system-state variables with respect to the model parameter values (Tomovic and Karplus, 1963; Gardner et al., 1981). A sensitivity analysis of a simulation model is conducted by measuring change in the deterministic model solution in response to changes in the model parameters. Such analyses also can be used with models containing stochastic elements by either disabling the random number generator and using the mean parameter values or choosing the same random sequence for each run in the analysis. The latter is easily done in most programming languages by making the randomization "seed" a constant.

Depending on the mathematical complexity of the model (i.e., if the model equations are especially few and continuous), the partials might be evaluated analytically. Models of greater complexity (i.e., virtually all models useful for risk assessment) necessitate numerical approximation of the partial derivatives. Gardner et al. (1981) present a comprehensive comparison of analytical and numerical sensitivity analyses of a lotic ecosystem model. Bartell et al. (1986a) compare individual parameter perturbation methods and Monte Carlo (i.e., simultaneous stochastic variation of groups of parameters) methods of sensitivity analysis of fish bioenergetics models.

Sensitivity analysis should not be confused with uncertainty analysis. Uncertainty analysis refers to the quantification of variability in model output when best estimates of realistic variability in model parameters and external forcing functions are propagated through the model. The methods of error propagation outlined earlier in this paper have all been used to assess model uncertainties (see table 9-2).

4.5.2. *Model Verification*

Verification is the step in model development where the mathematics and computer coding are examined to determine if the intentions of the modeler(s) have been accurately represented in the model construct. This step involves the detailed examination of the model to ascertain whether the functional formulations, external forcing functions, initial conditions, parameter values, and other model assumptions and constructs have been faithfully articulated in the coding of the model. The strongest verification is obtained by testing the model predictions against a data set from the system/site the model was designed to represent (Shugart and West, 1980). Successful verification of the model is a prelude to the basic execution of the model.

4.5.3. *Model Validation (Evaluation)*

Comparisons of model predictions with independent data sets from the same or other systems (depending on the generality expected of the model) are used to validate models (Shugart and West, 1980; Caswell, 1975). In developing the guidelines for ecological risk, we suggest replacing the term validation with *evaluation*. Models are simplifications of nature and, therefore, invalid in the sense that the model cannot predict more than some small set of the behavior of the real system. Any model can be pushed to the point of failure. Moreover, the model builder, intimate with the strengths and limitations of the model, can design any number of experiments that will support or refute the model. A model cannot be "validated" anymore than a statistical hypothesis can be proved. Model

utility, in the terminology of Mankin et al. (1975), should replace validation in method and concept. The emphasis should be placed on determining the conditions where model results are likely to be sufficiently accurate for useful risk estimation; that is, accurate enough to produce information conducive to coherent decision making. The degree of accuracy undoubtedly will be influenced by the severity of the consequences of an incorrect decision. Thus criteria concerning utility will vary by model and model application.

Model evaluation should proceed to a determination of its domain of applicability. The values of initial conditions, model parameters, and external forcing functions that prove conducive to accurate results should be determined for the model. Similarly, conditions that suggest likely inaccuracies in model results should be described. Refinements, guided by the results of sensitivity analysis, can be directed at increasing the domain of model applicability.

4.6. Natural Variability

Ecological risks will have to be evaluated in the context of natural variation in the endpoint of interest, since ecological systems are dynamic in space and time. Fluctuations in system state (e.g., species richness, population size, primary productivity, mineral cycling efficiencies, decomposition rates) occur in the absence of human disturbance. Therefore the accurate assessment of ecological risks in dynamic systems will depend on the characteristic variability of the selected system endpoints. Simply stated, the greater the natural fluctuations in an ecological endpoint, the greater the difficulty in measuring a response to disturbance; hence the greater the difficulty in assessing ecological risk.

Quantitative understanding of the characteristic fluctuations in ecological systems can be used to judiciously select endpoints for assessment. First, endpoints should be selected in terms of their dynamics and referenced to the decision-making or regulatory environment. For example, selection of changes in the composition of forest tree species as an endpoint in risk assessment is simply not practical in the context of regulatory frameworks with dominant 2- and 4-year political cycles, even though the ecological consequences could be highly significant. At the other end of the spectrum, for example, selection of changes in adenylate phosphate pools as an endpoint could require sampling at a scale of milliseconds—again impractical. The lesson to be learned: Choose endpoints that permit measurement of sufficient grain and extent (Allen and Starr, 1982) to conveniently characterize natural fluctuation and departures from natural fluctuations.

Knowledge of characteristic system dynamics also can be used to speculate where in an ecological system a disturbance might produce its greatest effect or highest risk. System components that

fluctuate naturally on scales similar to that of the disturbance are at risk. If the disturbance is extremely "slow," or low frequency, the system can adapt; an extremely high frequency disturbance simply may not be "seen" by the system (Allen and Starr, 1982). Thus known natural variability might be used to advantage in selecting endpoints for assessing a disturbance of known frequency or recurrence.

4.7. Multiple Stressors

A critical challenge to developing capabilities in risk characterization lies in recognizing that single, isolated stressors are seldom the focus of real-world ecological risks. Except for isolated spills (e.g., petroleum hydrocarbons) or intentional broad-scale applications (e.g., agricultural herbicides and biocides) of a single compound or narrow class of chemical congeners (e.g., PCBs), most environments are variously contaminated by a diverse assemblage of organic contaminants, fertilizers, trace metals, pesticides, and radionuclides. The ultimate ecological risks are determined by complex, interacting chemical mixtures. In the case of natural stresses, systems suffer at larger scales in space and time from fire, flood, drought, and pest outbreaks.

Assessments of multiple stressors will require establishing a scientific basis for choosing among alternative models for combined effects. In the case of toxic chemicals, the properties of additivity, antagonism, synergism, or domination by the most toxic chemical (DiToro et al., 1988) are alternative models that may apply for various mixtures. Current capabilities and experience in choosing a priori from these models for particular assessment are extremely limited. Experimental approaches to chemical mixtures have consisted of simple factorial designs using two or three compounds or the use of the mixture as a "chemical" in performing toxicity assays and conducting experiments in the laboratory (Franco et al., 1984) or field (Giddings et al., 1984).

Modeling approaches have emphasized the application of chemical speciation models (e.g., MINEQUL), assuming conditions of thermodynamic equilibrium, to calculate the degree of expected freely dissolved form, complexed compound, and different valence states (i.e., metals). These models are of unknown reliability when making calculations for chemical systems that depart from controlled laboratory situations.

4.8. System Resistance and Resilience

Measures of both the degree to which a system is sensitive to a stressor regime and the degree to which it might compensate or recover are covered in detail by other chapters (see chapters 5 and 7). Yet an understanding of the major components of this sensitivity is necessary in any discussion of risk integration methodology. Compensatory effects within the system are responsible for the degree of both resistance and resilience exhibited by the system in response to the perturbation induced by a stressor. Although the terms resistance and resilience sometimes have been used in conflicting ways in the ecological literature. One useful set of definitions follows the dictionary: Resistance is the inverse of the degree to which a system changes in response to a perturbation (i.e., the inverse of sensitivity), whereas resilience is a measure of the rate at which a system returns to the original state following the removal of the disturbance (Webster et al., 1975; Wade and Webster, 1976; Carpenter et al., 1992). Resistance also has been referred to as inertia (Westman, 1978), and a quite insightful but different use of resilience is that used by Holling (1986) in his discussion of how ecosystems are perceived and how they function (see section 3.2). Although resistance and resilience generally are measured only in systems (or models) that exhibit locally stable steady states, no theoretical reason argues against using them in the evaluation of systems and models exhibiting stable limit cycles. In such uses, however, the evaluation of resistance and resilience must be conducted in a more subjective, less formal manner.

Formal evaluation of both resistance and resilience is illustrated by Carpenter et al. (1992) using the methods found in Harrison and Fekete (1980) and DeAngelis et al. (1989b). These calculations can be made on data representing either measured flows in experiments on real systems or on models consisting of sets of linear differential equations. For models with nonlinear and often discontinuous differential equations, direct application of these techniques is not possible. In such cases, however, the model simulations are run, and the resulting flow data from any given simulation can be analyzed with respect to model resistance and resilience.

These attributes of systems provide important information about the potential response(s) of a system to perturbations. They are themselves dependent, however, on the type of stressor and its intensity, duration, and frequency. These are all characteristics that must be explicitly specified at the outset of model development if the goals of the model include predicting the resistance and resilience of the system.

4.9. Monitoring Variability, EMAP, and Ecological Risk Assessment

Spatial/temporal variability in systems is handled by standard sampling and statistical techniques. Although spatial variability may be treated in either a time-dependent or time- independent manner, temporal variability in ecosystems requires time-related data. For certain attributes of both populations and ecosystems, historical records of variability are available for physical environmental attributes (e.g., gas content of glacial vacuoles and oxygen isotopes in carbonate) as well as for some physiological processes (e.g., tree ring width). In most cases, however, contemporary long-term monitoring must be conducted.

4.9.1. *Monitoring Variability*

The degree to which the target system varies during the natural course of time is clearly an important component of risk assessment. If the degree of variability of the unstressed system is not known, the evaluation of predictions of the response to a stressor will be compromised. Indeed, many of the predicted responses to stressors may take years, decades, or even longer to manifest themselves. Yet once apparent, responses may be extremely serious. Consider the current situation with respect to atmospheric contaminants and the ozone layer. Predictions of such relatively slow (or at least long-lag) phenomena must be evaluated in the context of some framework that defines the normal variation around which a constant mean may be assumed. It is even more difficult to make predictions about the effects of a stressor on systems in which the unstressed condition is one of slow change upward or downward (e.g., sea level rise or warming/cooling trends).

Although any model can be simulated to provide a predicted rate of change in the mean of any given attribute, the evaluation of these predictions can only be performed through comparison with a set of long-term data. Yet long-term data sets are problematic, for two reasons: (1) there is resistance on the part of foundations and government agencies that fund scientific research to commit funds on a permanent or even semipermanent basis, and, in any case, most government agencies have funding "windows" that make it difficult to secure long-term support; and (2) long-term monitoring does not often result in interesting or counterintuitive breakthroughs, thus making its justification more difficult.

4.9.2. *EPA's EMAP and Ecological Risk Assessment*

The two difficulties listed above are gradually being overcome. On the one hand, there is growing recognition of the need not only to monitor the normal variability in the ecosystems of which

we are embedded but also the need to be able to provide an early warning when change in a critical state variable or parameter begins to exceed this "normal" rate of change (e.g., note the usage of the Mauna Loa carbon dioxide measurements).

A specific response to the need for long-term monitoring was conceived and implemented by EPA with EMAP (Olsen, 1992). The goal of this program is to identify environmental variables that will repay long-term monitoring throughout the United States by enabling detection of relatively subtle departures from their normal means and standard deviations. Thus EMAP is intended to serve as a tool for assessing the relative "health" of the environment and pinpointing areas and/or species that need attention—perhaps remedial, perhaps initially a more detailed assessment.

Although EMAP is not designed to predict risk, the data from its monitoring efforts should prove valuable for parameterizing some risk models and for testing model predictions about the dynamics of systems exhibiting "normal" behavior. We advise that the ecological risk assessment protocol not only plan to take advantage of the databases that will be provided by the EMAP initiative, but that means be found to inform the managers of EMAP about the general and specific needs of the risk assessment guidelines. In this way, as EMAP evolves and reassesses its own needs and goals, it can, wherever possible, incorporate the needs of this and other initiatives within EPA.

4.10. Recovery

The complement to ecological risk is recovery, which is defined as the system's response once the disturbance is removed. In a classical sense, recovery measures the return of the system to conditions that pertained before the disturbance.

Recovery is conceptually related to system stability, particularly to resilience (see section 4.8 and referenced chapter 7).

4.10.1. *Criteria for Assessing Recovery*

Recovery can be assessed according to various criteria. Following the general spirit of the guidelines, however, recovery should be assessed in terms of the response variables that were used to estimate or characterize ecological risk. If these variables return to their predisturbance values after removal of the disturbance, the system can be judged to have recovered.

Alternatively, other criteria can be chosen to monitor recovery. Similar to risk estimation, recovery can be assessed using population, community, or ecosystem measures. No recipe exists for selecting recovery criteria; recovery will be defined in relation to the risk endpoints and the

disturbance. The important point is that criteria must be chosen and their predisturbance values must be known.

4.10.2. *Exposure-Recovery Functions*

Exposure-recovery functions define relations between the degree of the disturbance (i.e., magnitude, frequency, and duration) and the time required for recovery. In essence, predicting recovery is the reverse of predicting a response to the stressor in the first place. All models of succession, for example, are models of recovery from the effects of some stress, whether acute or long term. Thus the same rules apply when constructing recovery models as when constructing risk models: If data on recoveries from differing magnitudes, frequencies, and durations of stress exist, then empirical models can be constructed; otherwise, mechanistic models must suffice as predictive tools.

4.10.3. *Probability of Recovery*

Realistic assessment of ecological recovery requires a probabilistic framework (Bartell et al., 1992). Given that ecological systems are characteristically dynamic, system states vary in space and time. A single system driven by seasonal cycles will demonstrate similar but not identical patterns of system state from year to year. A set of similar systems will exhibit similar but not identical behavior during the same period.

Long-term, systematic changes in state follow from the biotic-abiotic interactions that alter habitats and the physicochemical context. Natural variability and successional trends argue that recovery must be assessed against a backdrop of variance in the criteria used to measure recovery.

Variance also will characterize the postdisturbance system response. Spatial-temporal variations characteristic of the undisturbed system may be modified by the disturbance, or the disturbance might add another source of variance to the system.

The combination of natural system variability and variability in the disturbed system suggests that, at any point in space or time, meaningful comparisons of system states require the comparison of distributions. Recovery is an evaluation of the null hypothesis that the distributions of system state are identical over time or space. Thus recovery can never be assessed with complete certainty; rather, it can be best represented as a probability. Note that if the system stabilizes in a new configuration (i.e., alternative stable state), the probability of recovery according to the

predisturbance system state is zero; yet, according to criteria developed from stability theory, the system can be described as "recovered."

5. RISK SUMMARY

The selection and application of methods for risk characterization should take into consideration the form in which the risk analysis will be communicated to all concerned parties and, equally important, how the risk results will be used by risk managers.

The risk characterization is communicated through the risk summarization, which should facilitate the presentation of both qualitative assessments and quantitative risk estimates. The summary also should address the degree of confidence to be placed in the analysis as well as identify new information that could improve the assessment. The summary should finally assess the contribution of the risk characterization to the overall weight of evidence in the risk assessment. The following discussion identifies and discusses several important topics in each of the components of an effective risk summary.

5.1. Qualitative Versus Quantitative Assessment

Qualitative assessments of risk at their simplest are yes/no alternatives to the question of whether a stressor will prove to be a problem. On a more graduated scale, as we noted above, certain kinds of analyses can be considered qualitative (e.g., food web analysis) because sometimes they can give an ordered answer to the question.

Instead of considering the alternatives to qualitative versus quantitative assessment, we might find it more profitable to consider what level of measurement we can achieve with the chosen method for risk assessment. From the standpoint of the available statistical tests, the data from our analyses fall into the categories of nominal, ordinal, interval, and ratio (Siegel, 1956). Nominal data are purely classificatory (e.g., effect versus no effect). Note, however, that even here the yes or no probably will be based on some predicted attribute(s) of the system being "higher" or "lower" than "usual." Thus nominal data are generally of little interest in risk assessment, and the pure qualitative classification is also of relatively little interest. Ordinal data are what most classifiers have in mind when they talk of "qualitative" assessment (i.e., the effect is greater than or less than, but the amount of difference cannot be specified). The remaining two categories of data are the stuff of quantitative science. The interval scale of measurement requires that a measure of the difference between two ordered items be available, and the ratio level of categorization requires a true zero point. The vast majority of the quantitative data resulting from risk assessment will be of the latter type, since negative values for flows and standing stocks, for example, are of no relevance in ecology.

Thus most predictive methodologies of ecological risk assessment will be quantitative, with even the simplest initial question of yes or no generally being answered with data of at least the ordinal level.

5.2. Degree of Confidence

Confidence placed in the overall characterization of ecological risks for any application will be a function of the reliability of the information entering into the analysis and the competence of the integration and translation of this information into estimates of risk. Evaluation of the accuracy and utility of the risk characterization will require a thorough understanding of the quality of the exposure analysis and the exposure-response assessment. Additionally, an evaluation of the accuracy of the methods and models used to integrate this information must be performed, as well as an assessment of the assumptions and uncertainties that entered each stage of the overall risk analysis.

5.2.1. *Evaluation of the Exposure Analysis*

Evaluation of the exposure analysis should include an assessment of the scaling of the stress of concern, including its relevant frequency, magnitude, duration, and spatial extent. Was the exposure assessment based on site-specific measurements, differently scaled monitoring data, statistical extrapolations, analogy to other similar situations, or an educated guess? If a constant exposure was used in the analysis, was this justified in relation to what is generally known about the stressor of interest? If the stressor is a chemical, was the correct chemical species used to estimate exposure (e.g., PCB congener, metal valence state, polar or nonpolar aromatic molecule)? Answers to at least these questions are necessary to evaluate the accuracy and utility of the overall exposure assessment.

The exposure assessment is undoubtedly the central component of the risk characterization. The focus of the risk analysis is to translate the exposure estimate into estimates of its probable impacts on ecological resources. Imprecision in the estimated exposure is tolerable for extremely low or extremely high exposures; that is, in the "flatter" portions of the dose-response functions. Fundamentally, the steeper the slope in this function, the greater the implications of imprecision in exposure on the expected response. For situations where the best estimate of exposure lies in this region of the exposure-response function, risks should be estimated over the range of possible exposures, by using deterministic best- and worst-case exposures or by applying some error propagation method over an appropriate distribution of exposures.

5.2.2. *Evaluation of the Exposure-Response Relationships*

The quality of the exposure-response relationships used in the assessment should be summarized as part of the overall risk characterization. Several aspects of these functions can be addressed as part of the summary. First, it should be stated whether the stressor used to develop the functions was the same as the stressor of concern in the original problem formulation. For example, if a similar but not identical chemical species was used to develop the exposure-response functions in the assessment, this should be clearly noted. Substitution of analogous stressors, for whatever reason, can clearly introduce bias and uncertainty into the assessment.

Second, it will be unlikely that effects data will be routinely available for the same species of concern identified in the problem formulation. Thus the summary should address the nature and extent of species substitutions used to develop the exposure-response functions. The implications of using substitute species in terms of bias and uncertainty introduced to the assessment also should be discussed. The same argument applies in cases where the endpoint effect is not a species but a different taxonomic level of organization, or a community, or an ecosystem property (e.g., rate of decomposition or nutrient cycling).

Third, in assessments where exposure-response functions have been established between the exact stressor and endpoints of concern, the residual variance concerning these functions should be included in the risk characterization and presented in the risk summary. Graphical presentation of the functions and the supporting data may prove useful in this regard. Alternatively, confidence limits for the exposure-effects functions could be listed. The range of exposures over which the functions are expected to validly apply should be included in the risk summary.

Fourth, the nature of the extrapolation using the exposure-response functions should be outlined in the summary. Typically, the functions will have been developed from data collected in the laboratory or in other controlled conditions. In the assessment, these functions will necessarily be extrapolated to field circumstances defined in the problem formulation phase. The degree of departure between the scale of the assessment problem and the source of the exposure-response functions (i.e., somewhat analogous to the application-factor approach, see table 9-3) should be presented as part of the risk summary.

5.2.3. *Applicability of Methods and Models*

The methods of ecological risk analysis encompass a number of different kinds of models, and within each model category are a number of ways the model may be structured and employed. At a first approximation, we differentiate three classes of model: (1) empirical models, which require an a priori database on stressor/system responses; (2) process or mechanistic models, which are constructed on causal relationships; and (3) physical/experimental models, which are actual representations, at some scale, of the target system.

Empirical models may be constructed and manipulated using a variety of mathematical and statistical techniques; among these are single-value comparisons, joint distributions, regression models (of several types), fuzzy sets, and extreme event analysis. Empirical models are applicable for any stressor-response relationship that is within the range of the data set used to construct the model.

Process (mechanistic) models are generally classified by scale (i.e., as models focusing on the individual, population, community, ecosystem, and landscape or region). They may be used for prediction in situations where prior data on a range of stressor-response relationships are not available and where the nature of the stressor and/or the rarity or value of the target system precludes large-scale experimentation on examples of the target system. They are more data-demanding (and thus more expensive and difficult to construct) than are empirical models. Once verified, however, their predictions are less restricted than empirical models with respect to the range of stressor magnitude, frequency, and duration that may be simulated.

Physical models are generally classified in terms of scale as well, usually as either laboratory cosms of various sizes or large-scale field experiments. While cosms are physically bounded by the experimenter, field experiments usually are bounded by natural ecotones, although some of the large-bounded cosms are located in the natural environment. The results from experiments on physical models may be used to predict risk directly, or they may be used to implement empirical models. Data from physical models often are used to parameterize causal process models as well.

5.2.4. *Assumptions and Uncertainties*

The risk summary should detail the assumptions and uncertainties introduced in the course of overall assessment. As indicated in the previous sections, uncertainties can take the form of inaccuracy and imprecision in the estimates of exposure and the development of the exposure-response functions. Assumptions will necessarily be made in the implementation of models used to translate the exposure profile and the exposure-response functions into estimates of ecological risk. These

assumptions should be explicitly stated in the risk summary. Additionally, uncertainties in the form of gaps in information or scientific understanding (i.e., model uncertainty) can enter the assessment. These uncertainties should be summarized as part of an effective risk characterization (AIHC, 1992).

5.3. Contribution to Weight of Evidence

The risk characterization and the risk summary convey the results of the overall assessment to the risk managers. The ecological risks, however, represent only one component of an often complex decision-making process dictated by the particular legislation that mandated the assessment. Ecological risks must be of a form and content that can be used with other kinds of information (e.g., worker or human health risks, economic cost-benefit analyses, social implications, political considerations) in a comprehensive risk management or regulatory decision-making process. This concept of risks being part of an overall weight-of-evidence approach to decision making resulted to some degree from the concern that a single piece of "negative data" could dominate an analysis, while other "positive" aspects of a problem were ignored (Paustenbach, 1991). To be effective, therefore, the risk characterization need not be perfect. It should provide sufficient information, however, to allow the risk managers and other users of the assessment results to choose intelligently among the decision options (AIHC, 1992). Toward this end, the risk summary must accurately and concisely convey the ecological risks and, equally as important, address how reliable the risk estimates are in relation to the current scientific understanding and to the data, models, assumptions, and uncertainties that are part of the overall assessment.

5.4. Future Needs

The two major impediments to successful risk integration are the size and high cost of the database that underlies any model and the problems of organizing that database into a form (whether for an empirical or mechanistic model) that will yield accurate prediction with a minimum of uncertainty. As we argue above, risk decisions are increasingly involved with systems for which disruption is so potentially catastrophic that the shift in modeling is more and more to the a priori mechanistic model.

Advances in the technology of data loggers as well as remote sensors and microsensors will have a significant impact on the feasibility and cost of gathering the information necessary to construct predictive mechanistic models. Advances in the software (e.g., GIS) necessary for coding and manipulating remotely acquired databases will drive down the cost of constructing such models.

Increases in the complexity and veracity of such models, resulting from the likely continued rapid increase in computer memory and speed, will put a premium on the concurrent development of software expressly designed to visually present results in a manner conducive to the integration of the components of risk.

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Issue Paper
on

ASCERTAINING PUBLIC VALUES AFFECTING ECOLOGICAL RISK ASSESSMENT

Bryan G. Norton
School of Public Policy
Georgia Institute of Technology
Atlanta, Georgia

Prepared for:

Risk Assessment Forum
U.S. Environmental Protection Agency

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1. INTRODUCTION

Judgments of ecological significance cannot be based on purely descriptive criteria (see chapter 2, on ecological significance). Rather, since determinations regarding which ecological elements, structures, and processes are most important for society to protect inevitably rest upon value judgments, the ecological significance of a decision cannot be assessed except by reference to societal values. The decision of the U.S. Environmental Protection Agency (EPA) to focus on ecological as well as human health risk embodies, in itself, a decision to examine the impacts of human activities on systems, thereby implying that those systems merit protection. Systems have multiple scales, however, and the evaluation of ecological risk inevitably requires evaluations of which levels and scales of nature are of primary importance to humans.

While it is clear that ecological risk-based decision making requires input regarding public values, it is less clear how those values are to be determined, measured, and factored into decision making (Sagoff, 1988). Economists, ecologists, and philosophers pose value questions so differently that achievement of a single, consensually accepted methodology for ascertaining public values is unlikely at this time (Norton, in prep.).

The purposes of this chapter are to survey available methodologies for identifying and measuring societal values that may be at risk in decision making; to reveal underlying theories and discuss commitments that shape the data provided by relevant sciences; and to suggest practical guidance to those who must determine whether, and to what extent, a decision is likely to affect societal values. Given heated disagreement among practitioners of different disciplines regarding methods for ascertaining environmental values, the risk manager must use a balanced approach, drawing information from multiple sources and wisely choosing which value criterion and which value information to emphasize in particular situations.

Because mainstream economists have proposed what purports to be a comprehensive approach (at least in principle) to measuring societal values, our strategy is two-pronged. First, we summarize microeconomic (pricing) techniques espoused by mainstream economists and explore the rationale behind them. Then, appealing to criticisms by noneconomists and "ecological" economists, we show how the economists' calculations, while perhaps providing initial guidance, must be supplemented by other methods and by the good judgment of risk managers.

2. TECHNIQUES OF MAINSTREAM ECONOMIC ANALYSIS

Mainstream economists have generally attempted to determine public values associated with environmental goods through microeconomic analysis. They determine the value of a good or service by devising means to test how much consumers are willing to pay for it, given the competition of other demands on their assets. Because all resources are assumed to have adequate substitutes (in the sense that it is assumed that citizens will accept some amount of compensation for any particular loss of a means to satisfy some preference), economists can evaluate benefits enjoyed by individuals as a result of outputs of ecological systems and then aggregate these values toward a total. Economic methods, therefore, treat the problem of valuation as fully addressed in the processes of individual choices and behaviors. Economists believe that decisions (behaviors) of individuals seeking their own welfare provide, in principle, complete data regarding the value of any environmental good. Because many environmental goods and services are not traded in markets, the challenge for economists is to gather these data from individual behavior in ways other than direct observation of behaviors (see Freeman [1993] for a comprehensive treatment of economic methods to measure environmental and resource values).

Economists divide values into two categories: use values and nonuse values (Mitchell and Carson, 1989; Freeman, 1993). Some authors have also mentioned a separate category of "option" values (i.e., protecting something in case it is needed as a substitute for some currently used resource that may become scarce) (Randall, 1986; Fisher and Hanemann, 1985). Current thinking, however, treats option values as correction factors in the calculation of use values (Mitchell and Carson, 1989). Nonuse values are sometimes called "existence values" because they are values for protecting objects that are expressed by those who have no intention of using the object in question—they value the object for its existence alone, regardless of the use to which it can be put. For example, respondents may pay to protect wolves in Alaska, even though they have no intention of visiting Alaska. The value of an object, in this scheme, is the sum of its use values and its nonuse values. These categorizations are not particularly helpful in discussing intertemporal values because bequest values (i.e., a measure of the current generation's willingness to pay, altruistically, for goods to be experienced by future generations) apparently include both use and nonuse values. Judgments regarding protection for the future have generally singled out specific resources requiring protection, because economists have treated requirements of sustainability as questions in growth theory, which reduces sustainability to requiring an adequate investment policy (Solow, 1993). On this understanding of sustainability, each generation has an obligation to save at a rate that is "just,"

meaning a level of investment that provides future generations with the opportunity to achieve levels of personal welfare equal to, or larger than, its predecessors.

From the viewpoint of an ecologist, this conceptualization provides data that are too highly aggregated. This criterion of sustainability—which rests directly on the principle of intersubstitutability of resources—can only be quantified in terms that disregard distinctions between human-created and natural capital. This approach to intertemporal valuation rules out the identification of physical processes as worthy of protection for the benefits they provide—benefits that are impossible to specify in the present—to future generations.

3. MEASURES OF PUBLIC VALUES

Economists' methods for studying individual preferences can be categorized as follows: (1) direct market observation, which can be applied only to those objects that are actually traded in real markets; (2) extrapolation from market observation, which includes (a) direct observation plus adjustment for confounding variables such as government subsidies, and (b) hedonic pricing, which infers the value of objects that are not traded in markets from differentials in prices of other objects that are so traded (e.g., valuing the amenity of a beautiful view by observing real estate transactions and noting the price differential between otherwise similar properties with and without views); and (3) "contingent" valuation, which is designed to assess how much a person would be willing to pay for a nonmarket good or service if a market for it existed. This approach involves creating an artificial or hypothetical market as a means to reveal consumer preferences. Contingent valuation studies, in turn, may involve questionnaires, bidding games, and mock markets such as auctions; alternatively, it is possible to construct binding or nonbinding referenda regarding the imposition of a tax to accomplish the environmental goal.

Hypothetical markets, as an alternative measure, could assess how much a person would accept as just compensation for some good or amenity he or she currently possesses but may lose because of a public action or policy. Advocates of contingent valuation, however, have been reluctant to use figures derived from willingness-to-accept studies, because they have not fully explained why the willingness to accept the loss of a commodity consistently turns out to be significantly stronger than willingness to pay for the same commodity (Mitchell and Carson, 1989). A variety of explanations for this disparity have been proposed; economists usually bypass this problem by using willingness-to-pay figures as "lower-bound" or "most conservative" estimates.

When contingent valuation methods are used to discover prices for nonuse values (and there is usually no alternative to contingent valuation if dollar values are to be attached to nonuse values), then many further questions are raised. The willingness-to-pay figure assigned a nonuse "commodity" exists only theoretically. That is, in the nature of the case, no actual, measurable market behaviors can serve as a validation point for bids offered during a contingent valuation study of a nonuse value. This theoretical point is important because it means that preferences elicited in contingent valuation studies are a figment of the tests used to create them because the markets in which they are expressed are only the hypothetical markets constructed in questionnaires and the counter-to-fact scenarios they create. One cannot test the validity of these preferences against a pattern of real behaviors of individuals, but only against alternative tests with alternative scenarios. This is not to say that such

figures are useless, but only that they can be expected to be particularly context-dependent and highly volatile in relation to changing public priorities.

Contingent valuation studies are labor-intensive and expensive, and there can be pressure to transfer value figures, especially those that estimate nonuse values, from one situation to another. Because of the high level of contextual dependence of contingent valuations, the risk analysts must use utmost care if specific figures for willingness-to-pay figures from data taken in one locale and situation are applied in studies of other locales and situations. Thus, while examination of studies of similar situations can provide useful information, it is usually inaccurate to use willingness-to-pay figures derived by contingent valuation in one context in a cost-benefit study of actions in new locations.

A recent study by a blue-ribbon panel assembled by the National Oceanic and Atmospheric Administration (NOAA) cautiously endorsed contingent valuation as a method of estimating dollar figures for nonuse values, provided the studies follow prescribed guidelines. This conclusion, however, represented a qualified endorsement of contingent valuation for the specific purpose of proposing damage estimates to be used in court proceedings; it is not obvious that the NOAA panel's endorsement would also support use of contingent valuation data in a policy-setting process (NOAA, 1993). Even if, as the NOAA panel concluded, contingent valuation provides useful evidence regarding nonuse values in guiding damage assessments in the controlled situation of a courtroom setting, it does not follow that benefits figures can provide significant guidance in the more freewheeling context of environmental policy formation and evaluation. Further, the question of how much validity to accord hypothetical market valuations of environmental goods remains highly controversial: Many philosophers, ecologists, and dissenting economists question this entire theoretical framework for interpreting environmental values (see, e.g., Rolston, 1985, 1994; Kelman, 1981; Sagoff, 1988; Norton, 1988; Common et al., 1993).

One problem with applying contingent valuation methods to ecological risk must be emphasized here: Risk to ecological systems presumably includes risk to the "integrity" or "health" of these systems. Indeed, terms such as this appear in many pieces of legislation (Costanza et al., 1992; Rolston, 1994; Edwards and Regier, 1990). While the exact meaning of these terms remains controversial, proponents of these concepts agree that the terms must be defined relative to human and social values and that these characteristics emerge at the system level. Thus, to evaluate a change in an ecological system vis-à-vis the system's health or integrity necessarily requires assigning some value to changes in system-level states. This requirement places a tremendous burden on the

methodology of contingent valuation as used by economists because they assume that every change of system state that can be assigned a value must be associated with some change in the welfare states of human individuals or aggregations of individuals (Freeman, 1993). Since there are no acceptable methods for associating changes in ecosystem states with changes in individual welfare states, applying economic valuations of changes in ecosystem states is very difficult.

In principle, it is possible to design contingent valuation studies to estimate values associated with changes in the states of whole ecological systems; however, the difficulties of eliciting accurate estimates of willingness-to-pay figures for ecosystem-level protection are daunting. The research question would be: How much are consumers willing to pay to protect an ecological community or system such as a wetland or a bay (see Mullarkey and Bishop [1994] for an attempt to place values on wetland conversions). Here the role of context and information become overwhelming, and the contingent valuation technique is of questionable use. Those who emphasize methodology stress the importance of providing a strong information base so that the respondent in the study knows precisely the "commodity" that is being "purchased" in the hypothetical market. Because ecological relationships are usually not known precisely, it is impossible to develop a reliable characterization of the whole range of impacts on social welfare that can be expected from changes in ecological functioning as a result of policy choices, even if the policy choices are known. The survey researcher's inability to provide such detailed information can put the respondent in an unfamiliar situation, valuing an unfamiliar commodity and needing information unknown even to experts to express a grounded preference. While more exploratory research is needed in this area, it will be difficult to know how to interpret and evaluate contingent valuation studies of changes in ecosystem states until significant breakthroughs are made in determining how social welfare measures change in response to changes (i.e., degradation or restoration) in the state and functioning of ecological systems.

Another option to be explored is relaxation of the expectation that all benefits measures need to be aggregated together, allowing valuation information to take a variety of forms depending on which social values are at issue. This nonreductionistic approach holds out more hope for representing the whole range of human values. Adoption of moral pluralism is thought by some to increase the amount of subjectivity in the process (Callicott, 1990); however, a pluralistic approach can be integrated if sufficient attention is paid to the physical and temporal scale at which environmental problems are modeled and perceived scientifically (Norton, 1991).

Several motivations operate for seeking a more comprehensive, systematic method to evaluate management objectives; nevertheless, there is a foundational disagreement among policy analysts regarding how those objectives should be discussed and analyzed. One important motivation derives from the existing requirement—in legislation and international treaties—to manage ecological systems to protect their health or integrity. When environmental managers within EPA or other agencies of government wish to act to protect these whole-system characteristics, they face formidable challenges because of these methodological difficulties. A practical consequence for risk assessors, who would like to have as complete as possible an accounting of the benefits of proposed policies, is that justifying bottom-line decisions is difficult if information about values is offered in incommensurable forms. Nevertheless, if nonuse values and other difficult-to-quantify ecosystem-level values are to be included in benefits studies, these benefits must be ascertained by accepted methods or they will not be permitted by auditors at the U.S. Office of Management and Budget (OMB). These conflicting demands can lead to ecosystem effects being ignored. For example, in a recent National Academy of Sciences study, which attempted to weigh the relative costs of prevention versus accommodation to global climate change, researchers assigned zero value to damages to natural systems. The panel justified this exclusion by stating that no adequate methods for ecosystem valuation exist (NAS, 1992).

Thus, ecologists and environmentalists often complain that their most important points are ignored by economists; economists argue that ecologists have failed to show how ecological "risks" can be measured or how the goals of avoiding such risks should be assigned value and importance among the many competing social goals. Economists, on the one hand, and ecologists and environmental activists, on the other, tend to talk past each other, because no comprehensive language of policy analysis exists that can express these various positions impartially. In the next section, we suggest new concepts that may help to bridge the gulf between the two approaches and show how these concepts direct us toward new approaches to valuing environmental goods.

Since it is difficult or impossible to assign dollar values to preferences for ecosystem-level characteristics, replacement value has been suggested as a reasonable conceptualization of the societal values involved. Yet the approach is feasible only if one can identify those values and associated processes that the society will choose to perpetuate technologically if their natural sources are lost. This method should not be thought of as in direct competition with studies that ascertain the level of consumer preferences for a given commodity (willingness to pay). Indirect benefits emerge on the scale of the ecological system and in the intergenerational options available to a culture, and it is

difficult to make precise tradeoffs between a current willingness to pay and future benefits. Replacement value seems a more applicable concept *after a decision of ecological significance has already been reached.*

However useful in some contexts, replacement value will likely provide inflated figures as measurement of losses due to a policy or action. As economists have pointed out, a requirement of full-replacement analysis for any damaged resource assumes zero substitutability among resources. Thus, replacement value cannot be taken at face value, but must be corrected by some factor representative of the degree of substitutability of other resources for the damaged one. Once again, the problems with irreversibility, substitutability, and how to weigh these factors in decision making are central to important risk decisions. Moreover, because ecologists, economists, and others disagree regarding the level of substitutability in many decisions, decisions regarding how much to revise cost figures based on the appropriate degree of substitutability of other resources remains an important area for the judgments of risk professionals and managers. Further, it has been argued that judgments such as this probably cannot be decided based on any "one-size-fits-all" rule because the degree of substitutability is almost certainly highly dependent on the specific situations in which the rules are followed.

4. TOWARD A MORE "ECOLOGICAL" ECONOMICS

It is helpful to envision the intellectual terrain of risk assessment as represented in figure 10-1, which attempts to formulate issues separating ecological and mainstream economists in neutral terms (Norton, in prep.). This environmental risk decision square plots the variables of reversibility against the magnitude of impacts, defining a space in which various policy decisions can be located. Decisions that fit into this space are dependent on types of possible risks that are incurred in choosing among possible policies. As discussed in chapter 2 (on ecological significance) the decision square introduces spatial and temporal dimensions of risks that are important to ecologists and provides a formal representation of the rational decision space in which ecological risk assessments are made.[1]

Having introduced a formalized decision space that relates the differing positions outlined above, it is possible to add further assumptions and principles that give more structure to the decision space. This may provide more specific and substantive guidance in situations in which essential and irreplaceable resources are apparently at risk. First, we note that ecological economists, purely by virtue of their emphasis on ecological systems and processes, will focus on the temporal aspects of change. For them, irreversibility is not interpreted merely as an abstract concept of the substitutability of one resource for another, as measured against units of welfare available to consumers. Rather, it will include the more concrete parameter of reversal of impacts in ecological time and space. Thus, the horizontal axis can be calibrated as a measure of the time that would be required for natural processes to reverse a particular impact or disturbance. As such, the axis would locate decisions and policies that incur certain risks according to the recovery time needed if negative impacts occur as a result of that decision or policy.

From the ecological viewpoint, according to principles formalized within ecology as "hierarchy theory," this larger scale reflects an awareness that larger ecological and physical systems normally

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[1]It should be noted that the model, as here presented, is incomplete as a decision model because it does not incorporate a consideration of uncertainty/probability. Presumably, the obligation to act to avoid a future risk depends also on the degree of probability attached to the likelihood of the risk, given current knowledge that a negative outcome will occur. That is, if two risks are posed in the present, the negative outcomes of which would be equally harmful to an equal number of people, we assume there is a stronger obligation to act to avert a risk that has a 40 percent chance of negative outcome than to avert a risk that has a .05 percent likelihood of occurring. Work in this area is too speculative to guide policy at this time, however. For simplicity, we will use only versions that plot decisions in two dimensions. In effect, that means we are assuming that, in comparing two risks, the risks are equally likely to result in negative outcomes. If this assumption does not hold, risk managers should modify their judgments accordingly.

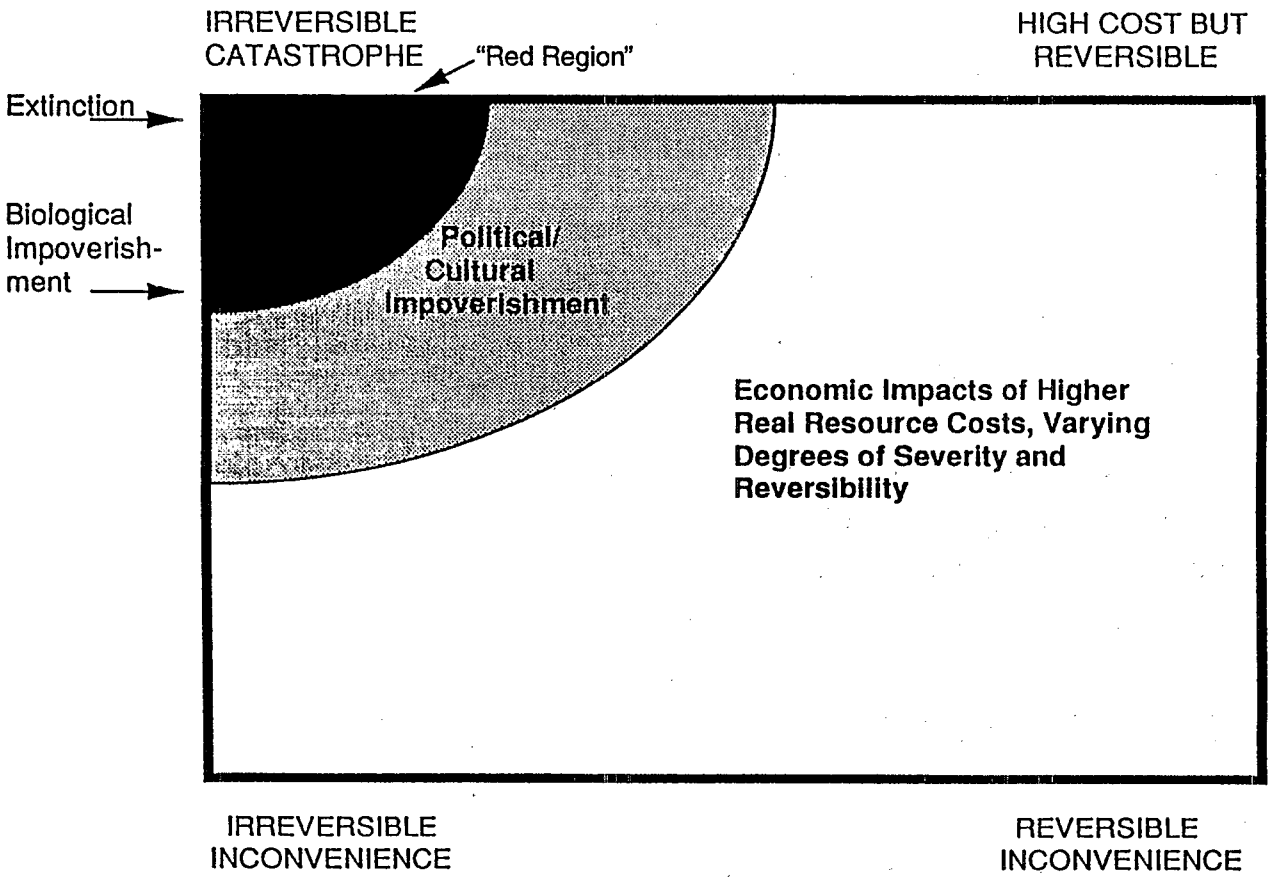

Figure 10-1. Risk typology: severity and reversibility (Norton, 1992).

change more slowly than their component parts (Allen and Starr, 1982; O'Neil, et al., 1986; Norton, 1991; Norton and Ulanowicz, 1992). The hierarchical approach has a profound impact on the way ecological economists view the relationship of economics to ecology because individual economic decisions are relatively short-term responses to evolving conditions in a hierarchy of larger and larger social systems (e.g., markets). In turn, these social processes are embedded in larger ecological systems and in even larger physical systems (e.g., the atmosphere). Thus, ecological economics accommodates the assumption that humans decide and act within a multilevel, complex system and do not always seek reduction of all values to a single measure that allows all values to be aggregated together.

On this basis, we can represent the claim, often heard in policy circles, that economists and ecologists employ a different "paradigm" (Daly and Cobb, 1989; Norton, 1992; Norton, in prep.). At the expense of losing the ability to aggregate fully across the whole space, ecological economists draw distinctions in types of risks, breaking the risk-decision space into distinct regions within which different considerations and criteria apply in different situations. It is argued in chapter 2 that the physical and temporal scale of impacts and the degree of substitutability of resources in a given situation determine the location of a particular decision in the decision space. Ecologists, therefore, differ from mainstream economists who assume that all types of risks can be expressed as dollar values that vary gradually and at the margins, representing their decision to treat all risks and benefits in "fungible" terms. Mainstream economists, ecological economists, and ecologists all agree that preservation of options for the future is desirable. Because mainstream economists believe there is a suitable substitute for every resource, however, they pay less attention to protection of particular resources that are at risk of irreversible damage (but see Krutilla, 1967; Krutilla and Fisher, 1975; for a discussion of the role of irreversibility in decisions affecting public goods). Ecological processes of restoration are not relevant to classifying risks in an economic analysis because reversibility is not considered the reversal of specific damage to a particular resource. Rather, reversibility is considered fully achieved by compensation in dollars sufficient to make the consumer willing to accept the damage.

In contrast, ecologists argue that ecological systems change discontinuously and that some changes (e.g., the extinction of a species or the destruction of a rainforest) introduce crucial thresholds into the decision process. In questioning whether there are "suitable substitutes" for, for example, functioning rainforests or for grizzly bears in Yellowstone Park, many ecologists urge—in

strongly moral terms—that extraordinary efforts to protect such resources are morally mandatory. In their view, therefore, such resources are not interchangeable with other possible losses or risks.

5. TWO-TIERED SYSTEMS OF ANALYSIS

Given these contrasts, we can begin to characterize a "two-tiered," or "hybrid," approach to environmental decision making that incorporates aspects of mainstream and ecological approaches into a more comprehensive system of valuation (Page, 1977; Page, 1991; Norton, 1991; Toman, 1992). This two-tiered approach is distinct from the single-tiered approach of mainstream economists that evaluates all values as present preferences. This approach would regard the decision space faced by environmental managers as split into regions, with the corner of the square characterized by important negative outcomes that are irreversible as representing an area of risk where values do not vary continuously with consumptive values and where moral strictures may apply. Thus, the two-tiered approach recognizes an important role for cost-benefit analyses expressed in dollar figures. Such analyses provide a useful method in decision areas where either reversibility is high or scale of impact is relatively small, or both. In these regions, we will be inclined to accept the usefulness of economic methods on the assumption that future generations cannot fault us if we compensate them for destructive impacts on natural systems with, for instance, increased technological know-how or monetary capital. In these regions of the decision space, in other words, decisions are regarded as fungible. In the "red region," however, social goals will be set by public discussion and debate (through normal political channels) and economists will contribute a cost-effectiveness analysis, ranking the various means by which to achieve the social goal. Economists thus could accept the same "geography" of the decision space, provided they agree that for some decisions moral limitations override and make irrelevant a computation of costs and benefits. This conceptual geography focuses attention on the "criterion of ecological significance"—a criterion that is based on physical parameters and social values, as explored in chapter 2.

In so dividing our decision space we have not violated our policy of conceptual neutrality, however, because the position of an economist who says that all decisions will be decided on the basis of a positive ratio of costs to benefits can represent his or her position by asserting that the "red region" is empty (Norton, in prep.). No decisions have a special ecological as well as a special economic significance. Thus, the position that all resources have substitutes represents the limiting case of not taking irreversibility or magnitude of impacts as having any nonfungible value. Ecologists differ from economists in believing that scalar differences can be so great as to introduce discontinuities into the analysis; economists rely entirely on a marginal, incrementalist style of analysis.

Relying on hierarchy theory and its principle that larger, supersystems change more slowly than smaller subsystems, we can calibrate the vertical axis of figure 10-1 according to the spatial scale of impacts, creating a grid that will represent decisions as located on the decision space. Risks that threaten harm that is prevalent over a large geographical area and irreversible for a very long time will be clustered in the "red region." We expect, given our model, that decisions in this region may be ecologically significant and may be governed by moral constraints. Decisions that risk small-scale and quickly reversible impacts are not significant on the ecological scale and fall in the area governed by economic reasoning. Applying such reasoning, Norton and Ulanowicz (1992), for example, have argued that, because a commitment to sustain biodiversity is consensually understood to be a commitment to do so for many generations of humans (at least for 150 years), we can conclude that the focus of biodiversity policy should be landscape-level ecosystems that normally change on a different temporal scale, an ecological scale that is slow relative to changes in economic behavior. If we can isolate the dynamics driving local economic opportunities from the dynamics supporting biodiversity, it may be possible to encourage both sets of dynamics and avoid policy gridlock, such as that which occurred over the Northern spotted owl in the Pacific Northwest.

The central principle of hierarchical organization deeply affects the valuation process as envisaged by ecological economists. Rather than aggregating values that are expressed on these multiple levels (which would require that the values be measured in a single metric, such as present dollars), ecological economists recognize that subsystems existing at different scales are driven by relatively distinct dynamics. Therefore, they expect humans to express and pursue many values (which are not reducible to a single common denominator such as the current willingness to pay), and they see their task as one of integrating multiple values (e.g., maintaining high levels of individual welfare and protecting resources for the future by protecting the integrity of ecological systems) rather than as reducing all values to a common metric and then aggregating toward a bottom-line ratio of costs and benefits as a guide to policy decisions.

Because crucial variables driving change at different levels of a system can vary considerably, independently of each other, it is possible in principle that important economic goals can be achieved (by manipulating variables affecting individual productivity and welfare) while environmental goods also are protected, or even enhanced. This will occur if policies can be devised to protect and enhance distinct variables affecting the environment at multiple levels, all of which support social values. Accomplishing this would require modeling the different changes on different scales and seeking to influence dynamics that are relatively independent across scales. To the extent that

ecological economists can identify dynamics driving economic opportunity (dynamics that are independent of the larger dynamic of ecosystem health), it may be possible to avoid choices between growth and the environment by providing economic incentives that will encourage new, cleaner, and more efficient technology. The problems associated with environmental management, then, come to involve separate dynamics associated with various public purposes, creating economically efficient means to encourage socially important dynamics (e.g., energy flows) and devising means to discourage damaging technologies by encouraging options for increasing productivity that incur less ecological damage.

Ecological economics thus recognizes another level, or scale, of decision making: the macroscale on which societies and cultures set conditions on market behavior and thereby influence individual decisions by changing the conditions within which individual decisions are made. It would be a mistake to assume, however, that ecological economists necessarily favor increased regulation. Rather, they favor experimental programs designed to increase economic opportunities and reverse environmental damage by protecting and restoring ecological processes essential to long-term, societal values such as sustainability of resources.

The central issue facing the risk manager can now be formulated quite simply: How many (if any) environmental decisions are located in the "red region" and thus governed by moral rather than economic, criteria? If there are any, the goal should be to determine precisely which decisions fall into these two broad categories. Judgments of ecological significance are a central component of this decision-making dynamic. Decisions that must be addressed in moral terms may represent decisions that have ecological significance because, given our two-tiered decision process, risks of ecologically significant changes will be exactly those that may affect large ecological systems for long periods of time. It stands to reason that risk managers should carefully scrutinize decisions of this type; it is plausible to expect that these types of decisions should be made according to somewhat different criteria.

A two-tiered system of analysis has the following structure: First, there are multiple criteria for good management that can be thought of as the first-order rules for decision making in actual contexts. To take a simple version, assume there are two first-order criteria—one of them is the standard cost-benefit calculation; the other might be a version of the Safe Minimum Standard of Conservation, which counsels that the resource in question should be saved, provided the social costs are bearable (Ciriacy-Wantrup, 1952; Bishop, 1978; Norton, 1987; Norton, 1991). This more stringent Safe Minimum Standard criterion shifts the burden of proof from the regulator (who is

obliged to cite adequate benefits to offset costs of a regulatory action) to the developer and advocate of growth (who must show that the costs of a protective effort will be unbearable). To decide which of these criteria to apply in a given situation, it would be necessary to invoke a second-order rule that guides managers based on contextual information. Thus, the important task in operationalizing any two-tiered system is to formulate second-order rules that are precise enough to give guidance in particular contexts and at the same time sufficiently flexible to allow essential managerial judgments of the appropriateness of first-order rules in a given situation. We have suggested (see chapter 2) that the spatial scale and the degree of irreversibility of impacts are important considerations in this second-order judgment.

It is likely that the further development of new methods for measuring natural capital and its depletion will remain an important preoccupation of economists who are sensitive to ecological issues for two reasons: first, they believe that a prerequisite to improving the treatment of natural systems is understanding and quantifying ways in which exploitative and careless treatment of the environment reduces opportunities for future and sustained growth; second, the problem of devising a new or expanded accounting system provides a forum for discussing the comparative strengths and weaknesses of economic and ecological methods of valuation. Accordingly, a central issue in devising alternative accounting methods is whether the new accounts will record dollar costs and benefits or whether there will be separate physical accounts, measured in terms of units of energy or other physical parameters (Odum, 1964; Hannon, 1991).[2]

A central methodological innovation proposed by a number of economists would supplement, or even replace, the use of the GNP as a measure of economic well-being (Repetto et al., 1982; Costanza, 1991). Some ecological economists propose methods of measuring increases and decreases in "natural capital" as a more realistic approach to keeping national accounts. These matters are highly controversial, however, and are related to problems of intersubstitutability of resources and the

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[2]It is useful to note that ecological economists and environmentalists attack the use of gross national product (GNP) as the key accounting framework on two separate grounds. First, they argue that GNP, which is a measure of economic activity, does not actually measure individual welfare. They point out, for example, that the *Exxon Valdez* oil spill resulted in huge cleanup expenditures that increased economic activity, but that these expenditures were indicative of losses, not gains, in individual welfare. Second, they note that economic growth as measured by GNP might indicate an increase in consumer welfare, but these increases might be offset (or partially offset) by associated declines in standing stocks of, for example, timber, or by degradation of ecological systems. In both of these cases, GNP is rendered inadequate as a guide to either individual welfare or economic "growth" in a broader sense that includes maintenance of natural capital.

degree of aggregation necessary and possible in measuring sustainability, as discussed above (see Solow, 1993).

These questions of evaluation and accounting are too difficult and contentious in nature to resolve fully here, though we contend that a more complex and comprehensive notion of public values is needed to address questions of ecological risk assessment and management. By juxtaposing societal against individual needs, intergenerational against economic time scales, landscapes against local spatial scales, nonuse (existence) against use values, and the concept of ecological reversibility against nonreversibility, a strategy might be devised for using the data and models of economists in conjunction with broader, ecologically based approaches to evaluation. This would provide an initial, if tentative, identification of societal values and incorporate them into judgments of ecological significance. As argued below, this tentative identification of social values and impacts on them may provide a concrete basis for discussion in an ongoing, iterative process of value articulation, experimentation, and revision.

6. RESOLUTION OF CONFLICTING VALUES

One approach to an effective environmental policy is to reduce value conflicts, whenever possible, by implementing policies that serve multiple values simultaneously. For example, if a wetlands mitigation project can improve hydrological function *and* provide habitat for experimental populations of an endangered species, without causing other environmental problems, it is clearly preferable to actions that pursue these goals separately. If, as hypothesized in chapter 2, the smaller-scale dynamics affecting individual decisions at the plot level are somewhat independent of the larger-scale dynamics that drive ecosystem health and integrity, it may be possible to integrate economic development and environmental protection. For example, development projects for countries of the South, such as planting woodlots for future fuel needs, can have positive economic, ecological, and long-term global impacts. Such projects present an opportunity to affect a variety of values expressed at the level of family welfare by providing sources of firewood as trees are thinned; at the level of ecological change, by reducing erosion and increasing water retention; and at the intergenerational and global level, by reducing birth rates. (Research shows that most families in less industrialized societies have more children because they need help with household chores such as gathering firewood and fetching water.)

To the extent that "lever" policies of this type can be devised and implemented, environmental policy can move beyond conflicts and direct regulations toward integration and positive programs of investment in natural capital. Once problems are properly formulated and modeled according to scale, and once all the public values in question are clearly articulated and associated with specific dynamics, policies with win-win or win-draw outcomes may become possible. This should be true, for example, in most or all of the cases in which risk assessors are able to determine that the assessed activity is ecologically insignificant because its impacts are easily reversible or because the scale of the impacts is small. In cases in which policies may affect ecological significance, creative thinking may lead to economic policies that enhance, rather than threaten, socially valued ecological processes.

In the end, irreducible conflict will remain, at least with respect to emphasis and priorities. Experts and interest groups will disagree among themselves over the judgment that certain activities entail risks in the "red region." There also will be disagreements regarding the strength of evidence of the likelihood of negative outcomes once outcomes are designated as possibly risky. Policy analysts probably will not agree on what information is relevant for deciding whether a given policy is risky.

7. LESSONS LEARNED FROM THE ENDANGERED SPECIES ACT

The Endangered Species Act (ESA) of 1973 and past attempts to interpret and enforce it provide managers with the best available experience in making public decisions regarding biological criteria. As eventually amended, that Act charges public servants to identify and protect all species that are "endangered" or "threatened," unless a high-level committee judges that the values of protecting a species are overridden by important national or regional interests. As such, implementation of the ESA serves as an example of agencies charged to act according to the criterion of the Safe Minimum Standard of Conservation, which requires society to protect resources threatened with irreversible loss, "provided the social costs are bearable." Note that this reasoning provides one example of decisions that are located in the "red region" on the basis of irreversibility. If one gives this event a spatial dimension by showing that the species and its services will be lost over an entire landscape, protection of endangered species can represent an example of decision processes that also can be located in the "red region" of the risk decision square.

The Act has had some remarkable successes, including, for example, its role in the recovery of the alligator in Florida and the Gulf Coast region. Of course, it also has sparked significant controversy, especially when protection of a species has been perceived to clash with important development projects.

On the positive side first, the ESA has clearly called public attention to the problem of biodiversity loss and therefore has made a huge contribution to educating the public about the importance and difficulty of protecting biological resources. In the process, the Act has served as a rallying point for environmentalists and also as a unifying target of criticism for advocates of economic development. Second, implementation of the Act has put in place a core of government scientists responsible for protecting biological diversity and resources as well as a set of bureaucratically specified procedures that are triggered when a species is listed as endangered or threatened. Third, the ESA has led to many successful cooperative efforts between the Federal government and State (and in some cases local) governments. Finally, research has shown that, for the vast majority of cases, the costs of protecting a species are remarkably low (Bishop, 1978; Tobin, 1990). On the whole, therefore, the ESA has succeeded to the extent that it has placed important issues on State and national agendas, raising public consciousness and stimulating public discussion and debate. In the vast majority of cases in which a jeopardy finding is issued, solutions acceptable to both the government agency and developers have been found.

Despite these considerable benefits, the Act has been far from an unqualified success. First, even when judged only against the internally stated objectives of the program, especially those of developing recovery programs for each endangered species, the effort has a low rate of success (Tobin, 1990). For a long time, this was a result of difficulties in designating critical habitats and in providing cost-benefit analyses sufficiently comprehensive to fulfill demands at the OMB. Some of these issues have been resolved, and there is reason to believe that a new and aggressive program will be developed within the U.S. Department of the Interior (U.S. DOI, 1994). Second, while protection of most species has cost little, there have been a few, highly publicized cases (e.g., involving the Northern spotted owl in the Pacific Northwest and the snail darter vs. the Tellico Dam) in which the Act has conflicted with forces favoring economic development. In those cases, the Act has been at the center of divisive debate, with opponents decrying it as placing more emphasis on birds and fish than on human welfare. Finally—and we argue below that this third weakness has contributed to the others—the Act has formulated debates about biological diversity and resources in an unfortunately individualistic, atomistic manner in which the target of protection is particular species or particular populations. Because wild species necessarily exist within complex ecological systems that form their habitats, it is doubtful that any policy approach that focuses mainly on species protection can adequately protect biological resources (Norton, 1986, 1987; Norton and Ulanowicz, 1992).

Important lessons applicable to the problem of defining ecological significance can be drawn from management experience with the ESA. First, the focus on species provides an understandable and communicable formulation of the problem of protecting biological resources, and it allows a fairly clear means to state goals for developing recovery programs for each species judged on biological grounds to be endangered. Nonetheless, the single-species formulation does not always present the issues involved in biological protection in the most perspicacious manner. In the spotted owl case, for example, the single-species formulation encouraged by the ESA reinforced the tendency for opponents of protection to cast the issue simply as a conflict between an owl species and jobs. In fact, of course, the conflict concerns use of old-growth forests much more broadly and has profound implications for other resources such as fisheries. The narrow, species-versus-jobs formulation stood in the way of public recognition that the real public choice was between short-term economic gains, on the one hand, and long-term protection of important ecosystem services and economic opportunities for deriving values from the remaining areas of old-growth forests, on the other. Thus, while the ESA is useful in situations where a species has become rare because of gradual habitat destruction and where a recovery plan with minor economic consequences can be devised, the Act is

less helpful in cases where a species becomes the crux of a broad debate about public goals and land-use planning over whole regions. Or, to put the same point differently, we have learned from the successes and failures of the ESA that, while the loss of a species is one indicator of irreversible ecosystem change and therefore raises questions of the "ecological significance" of those changes, the full consequences of those changes cannot be represented on only a single level of the ecosystem (e.g., species persistence).

To assess the full social and ecological significance of the loss of a single species or a population of a rare species, it is necessary to understand the particular species or population in its larger ecological context. Because ecological systems are best understood as hierarchically ordered, as noted above, changes in species composition must be characterized as one aspect of constant change taking place on many levels and scales. It is impossible to determine the ecological significance of a change in species composition without understanding the impact of that change on microprocesses, on other species' ability to persist in the area, and on the larger ecological processes that are manifest in ecological systems and across the larger landscape.

A second important lesson, therefore, emerges from the ESA: It is important to pay attention not just to the elements that perform ecological processes but, even more importantly, to the processes themselves. The ESA was in a sense the culmination of a long tradition of single-species, atomistic management that began with the monitoring and protection of game species more than a century ago. The rise of ecology and its emphasis on system processes first modified atomistic management by calling attention to the crucial role of habitat in supporting species. Even more recently, however, the definition of biological resources has been expanded to encompass "biological diversity," which includes not only species diversity and genetic variability within species, but also variability across habitats. Two populations of a species that exist in different ecological systems and niches, for example, are exposed to very different selection processes, which implies that diversity of ecosystem processes is crucial for the protection of species over multiple generations. More importantly, in the long run these processes are essential to ecosystem development and to evolutionary processes. If the long-range goal is to perpetuate diversity, then counting and protecting species is less important than understanding and protecting ecological processes within which species reproduce and adapt over time. One leading ecologist has built upon the insight that some species are "keystones" (i.e., species whose contributions are essential to maintain a stable ecosystem structure over time) and argued that we should be even more concerned about "keystone processes" (Holling, 1992).

Third, the public and academic discussions about how to enforce the Act have pointed up weaknesses in available valuation techniques as means to determine the full value of biological resources. In particular, the reductionistic methods of evaluating species for their impacts on human preferences (i.e., measuring their importance in enhancing human welfare) and then aggregating these measures have proved inadequate, at least in practice. While the concepts of "use value" and "existence value" may point in theory toward testable questions about human preferences that could in principle represent the "total dollar value of a species," the actual execution of such a benefits analysis has not been completed for a single species. Two reasons account for this: First, potential uses of a species are not ascertainable until exhaustive research is done on possible uses; for most species, it is impossible to ask a respondent to place a present dollar value on the future possible uses of a species when we know virtually nothing about the species or its members' life cycles. Second, even if the human uses of a species are identified and measured, no method has been developed for assessing the "contributory" value of species (i.e., the value a species has in developing and preserving ecosystem processes and in supporting other species that have direct and indirect impacts on human welfare through uses and through the enjoyment of free ecosystem services [Norton, 1988]). These discussions of how to value a species (and then how to factor that value into a cost-benefit decision-making process regarding whether, for example, to build a dam) have therefore clarified the limits of aggregative valuation processes when applied to ecosystem processes.

Finally, we can draw the lesson that, while a more ecological and less species-by-species approach requires a more complex formulation of the goals of protection programs, a proper formulation of the multileveled problem of assessing ecological significance can actually simplify the problem of protection. In some cases, for example, declining populations of a species in particular areas can be understood as a natural outcome of ecosystem development and be judged to be of no ecological significance. This judgment will be easier to make if the processes are understood and if the species is establishing and maintaining populations in other areas that are at different stages of ecological development. In certain cases, however, it might even be possible to conclude that loss of a species or population from a whole area is of little ecological significance. Making these latter conclusions will require an understanding of ecological processes, of course, but they will also require a well-developed public consensus about values and priorities in the protection of natural systems. Ultimately, these decisions will require some means to identify and articulate the social goals that are intimately intertwined with ecological processes. The progress made in understanding the strengths and weaknesses of the single-species approach as embodied in the ESA has led, at least indirectly, to

the discussions of ecosystem risk and to the struggle to articulate criteria of ecological significance (as exemplified by this chapter). One aspect of that project (see figure 10-1) is the determination of the correct scale on which to model natural processes and formulate environmental goals. To achieve the goal of protecting ecologically significant elements and processes in nature, scientists, managers, and the public must interact to articulate societal values, determine which dynamics are essential to protect those values, and formulate management goals.

8. TOWARD A MORE DYNAMIC AND COMPREHENSIVE VALUATION PROCESS

By adopting a two-tiered decision model, we have attempted to exhibit the importantly different roles of mainstream microeconomic analysis and of methods for identifying ecological risk. Since many environmental problems are problems of scale, we have argued that the basic structure of decision making should recognize at least a fuzzy line between those activities and policies that risk minor and reversible changes to the environment and those that risk changes that will affect whole ecological systems over long periods. In either case, this approach recognizes an important role for economics. In the former case, economic analysis may be given largely free rein; social choices, when not ecologically significant, can be governed by traditional economic analysis (corrected, we hope, for considerations of equity). In cases in which there is a determination that a risk of ecologically significant damage may result from an activity or policy, the role of economics is better seen as engaging in "cost-effectiveness analysis," which is an essential part of any application of the Safe Minimum Standard of Conservation. Whereas cost-benefit analysis balances all costs and benefits of a project, cost-effectiveness analysis accepts certain (e.g., politically mandated) constraints on goals and examines the various policy options available to achieve those goals, seeking the least costly method for achieving the mandated goal (Sagoff, 1988).

Several features of mainstream economic analysis, however, imply that this method has limits when applied to the question of whether an activity or policy carries risk of ecologically significant change. First, as noted above, economic analyses are seldom comprehensive because typically studies have not been completed to measure these values, even in cases where methods exist to do so. While it is useful to know, for example, that species may have option value, it is almost inconceivable that an "objective" figure could be given with any confidence to represent a rationally informed dollar figure for the option value of a species whose biological characteristics are hardly known. Second, as noted above (and in chapter 2, on ecological significance), ecological problems often involve discontinuities—activities we engage in may alter the trajectory of ecological change by passing thresholds beyond which stress to a system encourages rapid deterioration in the system's ability to fulfill human needs and values. Third, ecological significance is manifest not just on a single level, but on many levels. Ecosystem-level characteristics often govern the flow of ecological services; an emphasis on short-term individual welfare characteristic of economics is unlikely to reflect values that emerge on this scale. The second and third weaknesses of economic analyses have been addressed in this discussion by emphasizing the importance of understanding the temporal and spatial scale of risks encountered in the decision-making process.

Finally, economic analysis is not a particularly useful tool for determining values that emerge on the ecosystem level because the measurement of those values will be highly sensitive to the particular decision context. Any study of public preferences will be conducted at a particular time and be administered against a particular backdrop of information (which may be primarily the respondents' background knowledge or may rely more heavily on informational packets provided by researchers). Having noted above that preferences, especially of nonuse values, are theoretical entities with no referent in actual human behavior, this contextuality is unavoidable. This characteristic makes it especially difficult to transfer willingness-to-pay data from one context to another. It also follows that the way a question is framed (i.e., as a political, legal, or economic one) will affect the values that individuals express (Sagoff, 1988; Common et al., 1993). This preliminary research suggests that citizens might respond quite differently to a contingent valuation, such as a binding referendum on their willingness to pay taxes to protect parks and reserves, than they would when the question is posed as a matter of economic choices of individual consumers. Toman (1992) has explored the possibility that some questions of protection values are best considered by making an analogy to decisions made at a constitutional convention.

Four new trends in risk evaluational studies should be kept in mind: (1) the use of more interactive processes of evaluation and management; (2) an emphasis on locally based values; (3) an emphasis on interpretations of the scale of an activity as an important precursor to determining the values affected; and (4) a willingness, in cases where human values clearly depend on specific ongoing ecological processes, to conceptualize the value of an ecological resource as the cost that would be incurred to replace that process or function by artificial means.

Use of interactive processes. Whereas questionnaires can often provide important base-line data regarding social preferences, respondents' answers necessarily depend on the amount and quality of information at their disposal. In areas of uncertainty about both values and science, it is useful to engage the public in an ongoing process in which their evaluations lead to ecological models, and the models allow the proposal of more precise management goals. The use of focus groups or town meetings, for instance, as well as pilot projects in which communities "experiment" to determine if they like the outcome of management initiatives, may become more important in environmental value assessment. These ongoing processes of value articulation, discussion, and revision should be designed to maximize information feedback regarding the impacts of human actions and choices.

These processes will lead to progress only if the scientists, the public, and decision makers participate in an ongoing dialogue that is open and informative (Lee, 1993).

Emphasis on local values. Increasing emphasis in the literature is on the importance of particular place-oriented values (Ehrenfeld, 1993; Norton, 1991) in the protection of cultural connections with natural systems. For example, one proposed study will use focus groups made up of citizens who live and work on Chesapeake Bay to discuss the values they place on the bay and the goals they would adopt for managing the bay's ecosystem.

Interpretations of scale. An emphasis on the scale of economic activities and their impacts on ecological functioning focuses attention on diverse development opportunities that promise a positive, collective economic benefit but tend to cancel each other out in relation to the whole ecosystem. Assume, for example, that in a given area a trend toward monocultural agriculture is deemed to threaten the diversity of the landscape, and regional planning results in a shared agreement to diversify land use. Once the problem and goal are clearly articulated, it may be possible to make public investments (e.g., the establishment of farmers' markets in urban areas within the matrix) that will encourage some farmers to choose smaller-scale production of foodstuffs (e.g., vegetables, fruit orchards, honey production) for local markets. Integrative environmental policy, therefore, can be creative and enabling, rather than simply based on regulation.

Conceptualization of replacement impacts. Ecological properties that provide benefits to biological species are those structural characteristics (capital) that ensure sufficient survivorship of individuals and persistence of species. The structural properties of ecosystems both reflect and support species diversity, spatial complementarity, functional redundancy, and adequate proportionalities of functional niches (e.g., predator:prey ratios). If the structural integrity is maintained, then the critical functions are sustainable. These functions constitute "free goods" for human society and are generally taken for granted when short-term tradeoffs are addressed. For example, consider a typical pothole wetland with its complement of cattails, bacteria, algae, and animal fauna. The direct benefits (table 10-1) can be defined as those that result in the transport and fate of materials that are considered to be associated with environmental damages. Emergent vegetation has the highest productivity—anaerobic sediments store carbon dioxide and allow microbes to detoxify pesticides. One can calculate the cost of building a "bioreactor" or a "chemical reactor"

Table 10-1. The Direct and Indirect Benefits From Wetlands That Must Be Replaced in Volume, Space, and Kind for Economic Valuation of Natural Systems

Direct Benefits	Indirect Benefits
Denitrify nitrates	Biodiversity
Uptake of phosphorus	High productivity
Detoxify pesticides	Landscape diversification
Carbon dioxide sink	Breeding and nursery areas
Sequester eroded soils	
Modulate storm impacts	

that would perform the same function in the same volume at the same location. The wetland runs on solar energy (free goods) and regulates its own internal structure and rates of chemical processing. The engineered alternative would most likely run on fossil fuels, and the control system would probably be complex and expensive (i.e., you pay an engineer more than you do bacteria). The indirect benefits are those for which no easily engineered alternatives exist (see chapter 2, on ecological significance). The benefits of biodiversity and landscape diversification are generally recognized and are reflected in real-estate valuation. The high productivity (biomass energy) of breeding and nursery areas (fish and game recruitment) produce indirect benefits, since they could support other, primarily economic, activities.

Indirect benefits are often not registered in economic analyses that measure individual preferences because the values are most evident on the level at which the social community interacts with the larger landscape.

9. RISK AND UNCERTAINTY IN PROTECTING PUBLIC VALUES

What options are available to minimize and mitigate risk in those cases where it is determined that there is a significant possibility that there is ecologically and socially significant risk attendant on some human activity? The danger, one might argue, is that many activities, including activities that are necessary to fulfill basic human needs, may have some risk of harming ecological systems. Policy-makers may face the following quandary: There is significant risk that activity A will have negative consequences for ecological functioning. If the policy-maker takes the risk to be too high, activity A will cease, and the values derived from it will be lost to society; if the decision maker decides the risk is acceptable and activity A is allowed, then the government will be blamed if activity A leads to loss of welfare. The decision seems to hinge on the probability of the risk as well as the decision maker's attitude toward the risk. Some of this anxiety can be alleviated if one recognizes that activity A may be modified—that there may be alternative, lower-risk actions that will achieve the societal values associated with activity A. The risk manager, nevertheless, must (hopefully with democratic guidance) decide how much risk is too much.

It is also possible to spread risk around, directing some risk at the private sector, through the technique of "assurance bonding" (Costanza and Perrings, 1990).[3] According to this technique, private developers and other users of a resource would be required by a government agency to post a bond in the amount of a "worst-case" scenario for destruction of societal values. The burden would then be on the resource user to (a) show that its planned activities carry less risk, in which case the bond amount would be reduced, or (b) post the bond and, after work is completed, demonstrate that the social costs did not occur in order to receive a refund of the bond. This and other techniques may prove useful in cases where the risks of proposed activities cannot be estimated accurately.

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[3]This technique would in most cases require enabling legislation.

10. CONCLUSIONS AND GENERAL GUIDANCE FOR RISK PROFESSIONALS

Until recently, the risk assessment concept and the associated paradigm has been mainly concerned with individual health and has rested on a basic-value commitment to maximizing social welfare measured as the collectively aggregated well-being of individuals in society. Work in this area can usually proceed to judgments regarding the most efficient means to reduce risk, because everyone agrees that a reduction of risk will correspond to an increment of human welfare. The expansion of the concept of risk assessment to include risk of ecological damage, in the present or at some time in the future, will require a significant overhaul of the original paradigm because no agreed-upon methods exist by which to connect measures of changes in states of ecosystems (indicators) with measures of individual welfare. As a result of this serious gap in the science of risk analysis, it is imperative that risk assessors and environmental policy analysts address the problems of ecological accounting. This is an area where funding has increased, but far more must be done.

This chapter must be read in the context of the position taken in chapter 2 (on ecological significance), which argues that risk assessors must take social values into account at every stage of the risk assessment/risk management process and that judgments of ecological significance must pass a dual test. To be ecologically significant, a change must exceed the natural variability of the system for that parameter *and* involve impacts affecting significant social values. In chapter 2 and again here, we conclude that risk assessment and risk management that is sensitive to human values and to ecological information must involve an iterative process for formulating values in conjunction with scientific model development. This can only occur if there is ongoing dialogue and interchange of information among scientists, policy-makers, and the public. Moreover, the proposal, articulation, criticism, and revision of values and goals must pervade the whole process. This chapter simply highlights the process of value articulation and formulation, attempts to clarify the role of policy-makers and their judgments in this complex process, and suggests some general guidelines for the use of available tools to understand and protect important social values.

We have suggested two general techniques for measuring social values of changes in ecosystem processes and states. While not comprehensive, these processes can provide some quantitative methods by which to assign dollar figures to values associated with whole ecological systems. Quantitative methods can be used along with the iterative process between managers, scientists, and the public laid out here. One method is that of "replacement value," which many ecologists favor; however, this method systematically inflates benefits of protection unless substitutability appropriate to the case at hand is factored in. Analyses that attempt to estimate consumers' willingness to pay will

tend to underestimate the benefits of the protection of ecological processes because consumers (and even scientists) will be unaware of the importance of many beneficial functions. Managers might try to estimate the replacement value for all ecosystem services and other values, including nonuse values, derived from the ecosystem in question *and* the collective willingness of consumers to pay for protection, given competing needs and preferences. Assuming that the former figures will be significantly higher than the latter, managers can consider these figures as upper and lower bounds, respectively. They can then "adjust" these figures by favoring the replacement-value figures if the resource in question is scarce and has few low-cost substitutes or by favoring willingness-to-pay figures if the resource is plentiful or its functions are easily replaced.

Since no precise standard for making decisions of this type has been established, it is inevitable that the decision of which value calculation to weigh most heavily will necessarily be left to the risk manager. To make this decision, the risk manager will need as much comprehensive data on expected impacts as is available. Information regarding the likelihood that a resource, if degraded, can be replaced with ready substitutes is also important in making this judgment. All data must be organized carefully, according to temporal and spatial scale of impacts, and the values component of this process must have a scalar aspect for associating social values with the ecological dynamics that support them. The articulation of these values, and more concrete goals based upon them, must become a more interactive and iterative process, involving scientists, the public, and policy-makers in a process of dialogue, experimentation, and adjustment. This interaction, and the social valuations that emerge from it, should inform every stage of the risk assessment and risk management process.

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