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Application of Watershed Ecological Risk Assessment Methods to Watershed Management

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DISCLAIMER

ABSTRACT

Watersheds are frequently used to study and manage environmental resources because hydrologic boundaries define the flow of contaminants and other stressors. It is a challenge to incorporate scientific information in watershed management and planning. Ecological assessments of watersheds are complex because watersheds typically overlap multiple jurisdictional boundaries, are subjected to multiple environmental stressors, and have multiple stakeholders with diverse environmental and socioeconomic interests. Ecological risk assessment (ERA) is an approach that has successfully been used to increase the use of ecological science in decision making, by evaluating the likelihood that adverse ecological effects may result from exposure to one or more stressors, yet its application to watershed assessment is limited. The purpose of this report is to provide suggestions and examples for making scientific information more relevant to the needs of watershed managers by using ERA principles to help structure ecological assessments of watersheds.

This report supplements the *Guidelines for Ecological Risk Assessment* (U.S. EPA 1998a) by addressing issues commonly encountered when conducting watershed ecological assessments. Suggestions and examples to follow are provided based upon lessons learned from prior watershed ERAs. This report is of potential use to ecologists, hydrologists, watershed managers, risk assessors, landscape ecologists, and other scientists and managers seeking to increase the use of environmental assessment data in decision making.

Each activity and phase of the watershed ERA process is explained sequentially in this report. Guidance on how to involve stakeholders to generate environmental management goals and objectives is provided. The processes for selecting assessment endpoints, developing conceptual models, and selecting the exposure and effects pathways to be analyzed are described. Suggestions for predicting how multiple sources and stressors affect assessment endpoints are also provided; these include using multivariate analyses to compare land use with biotic measurements. In addition, the report suggests how to estimate, describe, and communicate risk and how to evaluate management alternatives.

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LIST OF ABBREVIATIONS AND ACRONYMS

BPJ	Best professional judgment
BMP	Best management practices
CADDIS	Causal Analysis/Diagnosis Decision Information System
COLD	Conditions suitable to support coldwater ecosystems
CWA	Clean Water Act
CWAP	Clean Water Action Plan
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
ERA	Ecological risk assessment
ESA	Endangered Species Act
EPT	Ephemeroptera, Plecopera, and Trichoptera
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FWS	U.S. Fish and Wildlife Service
GIS	Geographic information system
HSI	Habitat Suitability Index
IBI	Index of Biotic Integrity
ICI	Index of Community Integrity
LOE	Lines of evidence
LIT	Literature search
Miwb	Modified Index of Well-Being
MOS	Margin of safety
NCEA	National Center for Environmental Assessment
NEPA	National Environmental Policy Act
NRC	National Research Council
RRM	Relative Risk Model
SAB	Science Advisory Board
SIE	Stressor Identification Evaluation
TMDL	Total maximum daily load
TNC	The Nature Conservancy
TSCA	Toxic Substances Control Act
TVA	Tennessee Valley Authority
UAA	Use Attainability Analysis
WQS	Water quality standards

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1. EXECUTIVE SUMMARY

Environmental managers are increasingly using a watershed approach for making environmental decisions. These environmental managers include state, federal, or local regulatory or resource agency staff; local and county officials; watershed associations or councils; and river watch citizen groups. Many environmental decisions are made with much uncertainty, and the quality of such decisions would increase if more science were used in the decision making process. Ecological risk assessment (ERA) is a process to collect, organize, and analyze scientific information in order to evaluate the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (U.S. EPA, 1998a). This document discusses the merits of applying ERA principles to watershed management and describes how more science can be used in decision making.

The watershed approach is a framework for coordinating environmental management that focuses public- and private-sector efforts on addressing the highest priority problems within a hydrologically-defined geographic area (U.S. EPA, 1996). Watershed ERA integrates ERA methodologies with the watershed approach to improve the use of environmental monitoring and assessment data in watershed and regional decision making. This document is written primarily to provide guidance and examples for ecological risk assessors, scientists, and managers performing ecological assessments of watersheds. Some of the principles also should be useful to landscape or seascape ecologists performing spatial-scale environmental assessments and to others seeking to increase the use of ecological science in decision making. The document is structured around the elements of watershed ERA (planning, problem formulation, risk analysis, risk characterization, risk communication, and risk management) and is based primarily on lessons learned from EPA-sponsored case studies.

Watershed ERAs are complex because a watershed typically overlaps multiple jurisdictional boundaries and has multiple stakeholders with diverse environmental and socioeconomic interests. Planning is essential in ensuring that watershed ERAs address locally based socioeconomic and scientific challenges. Key elements of planning include identifying and involving relevant stakeholders, developing environmental management goals and objectives, and agreeing on the focus, scope, and complexity of an assessment. Appropriate goals and objectives should be based on existing watershed management plans, environmental organizations' mission statements, survey results, and other stakeholder opinion data. An interdisciplinary team of scientists and managers who understand ecological principles and possess local ecological and socioeconomic knowledge is needed to develop a set of scientific assessment objectives and management options.

During watershed assessments, the needs of managers and stakeholders may change and managers may need to take action before an assessment is completed. Thus, flexibility on how the ERA process is implemented, along with regular and recurring interactions between scientists and managers throughout the assessment process, are essential. ERA, and especially watershed ERA, is an iterative process. It is unlikely that everything will proceed precisely in the chronological manner presented in this report.

Watershed ERA problem formulation provides an organizing framework for the entire assessment. Assessment endpoints are identified and conceptual models and an analysis plan are developed. Assessment endpoint selection translates abstract environmental management objectives into specific, well-defined, and identifiable attributes of the system. Establishing a linkage between abstract management objectives and specific system attributes is particularly important in watershed assessments because of the ultimate need to choose between competing management objectives. Several assessment endpoints may be identified, but only a subset of them may be analyzed because resources, data availability, or realistic management control options may be limited. To select a subset of assessment endpoints to be analyzed, an assessment team composed of risk assessors, scientists, engineers, policy analysts, managers, and other professionals may use their best professional judgment to assess stressor impact and risk reduction opportunities.

Conceptual models are especially valuable in watershed assessments because they describe multiple physical, chemical, and biological stressors, their sources, and the pathways by which they affect various valued ecological resources. Due to the large number of pathways to be considered, the conceptual model can make it easier to identify causal pathways and establish priorities for analytical efforts. The model may also evolve as the assessment progresses and a better understanding of pathways and impacts is acquired.

The final product of problem formulation is an analysis plan documenting the proposed approach for the assessment. Sometimes, when assessment resources are limited or a decision must be made quickly, it may not be practical to complete a quantitative and formal assessment. In these instances, the qualitative information from problem formulation can be used for decision making.

Assuming the assessment team proceeds with the ERA, the next step is the risk analysis phase. It is ideal to develop stressor-response relationships that relate the magnitude, duration, frequency, and timing of exposure to the biological effects. However, generating such relationships for multiple interacting sources, stressors, and endpoints for a watershed assessment is not trivial. Geospatial analysis using geographic information system (GIS) tools usually play a pivotal role in analyzing land use, land cover, and other stressor and response data. To be cost effective and efficient, a watershed assessment might need to focus on assessing the relationships of one or a few major stressors (or their sources) on a small set of ecological effects rather than on attempting to quantify each stressor's exposure and effect on each endpoint. For instance, the focus could be on studying the relationship between percentage of urban land and fish species richness. Another approach is to examine relationships between groups of stressors, habitat data, and impact data (i.e., groups of responses). High relative risk occurs in the spatial areas where source, habitat, and impact co-occur. Categorical data or scores from one to five may be used rather than numerically modeled or predictive estimates. Moreover, it may be necessary to extrapolate results from a similar watershed as long as uncertainties are adequately documented. When data on the assessment endpoint are not available, a measure of effect on a surrogate may be commonly used instead. Finally, exposure and effects data may need to be aggregated, thus implying that the typical risk analysis phase is performed in conjunction with the next phase—risk characterization.

ERA risk characterization consists of risk estimation and risk description. Risk estimation is a difficult task because stressors can affect valued ecological resources in different habitats via multiple interacting pathways. The exposure and effects characterizations (or their aggregated estimates, as may be necessary in a watershed context) developed in the prior phase should be used as lines of evidence to support particular risk estimates. Lines of evidence also may include models, field work, and analyses performed elsewhere. All the lines of evidence need to be brought together to reach a final conclusion about the likelihood and the consequences of effects. The spatial distribution of exposure and effect is important to consider and is often assessed and presented using GIS data layers.

Risk description summarizes and presents information so that choices among alternative courses of action can be made with knowledge of the outcomes. Descriptions can be qualitative statements such as "option A causes more risk than option B." Effects data also can be color-coded onto maps of the watershed, allowing stakeholders to see potential impacts in the lakes and rivers that matter most to them. A data table can be used to show the various lines of evidence and to summarize the conclusions of risk and recovery potential to various life stages. Risk assessments need to be peer reviewed to help convince skeptics. Assumptions and uncertainty can be summarized in data tables or described in narrative form.

Successful communication of risk assessment findings requires that communication occur throughout the assessment process to ensure the results reflect the management objectives, decision context, and environmental conditions that exist at the time the information is needed. Managers need to tell assessors what they need, and scientists need to describe what they can provide. Effective risk communication must accurately translate the best available and most useful scientific information in a manner understandable to managers and stakeholders. Depending on the audience, the risk assessment information may need to be provided in a nontechnical manner. Clear presentation of results (e.g., maps, figures with simple dose-response curves) will enhance understanding of findings and foster consensus among stakeholders.

Watershed ERA principles can help watershed managers make more informed total maximum daily load (TMDL) decisions or take a wiser approach to identify or attain beneficial use designations. ERA principles can also assist managers in resource planning, making land use zoning decisions, and implementing best management practices (BMPs).

2. INTRODUCTION

This report discusses the merit of applying ERA principles to watershed management and describes how science can be better integrated into decision making. Environmental managers are increasingly using a watershed approach for making decisions. Environmental managers include regulatory or resource agency (state, federal, local) staff and any party with authority to implement a management plan (e.g., local and county officials with zoning oversight), watershed associations or councils, and river watch citizen groups. Many environmental management decisions are based on uncertain information and would benefit from having a stronger scientific basis.

ERA is a process to collect, organize, and analyze scientific information in order to evaluate the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (U.S. EPA, 1998a). ERA principles are relevant to environmental management because even though most decisions are made without a risk assessment, the decisions are fundamentally about managing ecological risks.

EPA and others, through various regulations, have reduced the impact of point source pollution and it is now widely recognized that further substantial improvements in the quality of U.S. lakes, streams, rivers, creeks, and ponds depend on controlling nonpoint source pollution (U.S. EPA, 2006). Nonpoint source pollution problems are not as well addressed by existing regulations because of the difficulties in identifying diffuse sources and ill-defined pathways. Efforts to address nonpoint source pollution rely heavily on voluntary compliance, stakeholder involvement, and a scientific understanding of the cumulative impacts of multiple physical, chemical, and biological stressors over a broad range of spatial scales.

Consistently incorporating science in watershed management decisions, however, is challenging. Multiple physical, chemical, and biological stressors result from human activities. These stressors, when combined with a network of interrelated environmental conditions, cause diverse impacts on numerous ecological resources. Tradeoffs among environmental, political, economic, and social factors often result from subjective value judgments that occur as part of the decision process. It is sometimes difficult to implement scientifically supportable actions because of conflicting interests, lack of certainty (or public trust) in the science, and poor communication of the risks and benefits, particularly to those who must face the risk. As a result of these challenges, data from many monitoring and assessment efforts frequently do not play a major role in management decisions (Ward, 1996; Ward et al., 1986).

Helpful suggestions and frameworks have been developed to increase the use of science in watershed and regional environmental management (Rhoads et al., 1999; Maxwell, 1998; U.S. EPA, 1996; Armitage, 1995; MacDonald, 1994; Slocombe, 1993; Ward et al., 1986). *Guidelines* *for Ecological Risk Assessment* (U.S. EPA, 1998a), written for all types of ecological risk assessment, generally addresses the application of ERA to watershed and regional assessment. Yet the application of ERA to these larger scales is still somewhat limited (Serveiss, 2002; U.S. EPA, 2000a). This report supplements the ERA guidelines by providing additional guidance for those performing watershed and regional ecological assessments.

In the mid-1990s, EPA's Risk Assessment Forum and Office of Water cosponsored the development of five demonstration watershed ERAs to test application of the ERA Framework (U.S. EPA, 1992) to Office of Water programs. The five watershed assessments were performed in the Clinch and Powell Valley, VA, Middle Snake River, ID, Waquoit Bay, MA, Big Darby Creek, OH, and the Middle [segment of the] Platte River, NE. All five selected sites had valued ecological resources, multiple stressors, an existing data set, and willing assessment participants. Using these assessments, many key aspects of the watershed ERA process, including challenges encountered and lessons learned, were documented. Several researchers (Diamond et al., 2002; Serveiss, 2002; U.S. EPA, 2000b; Butcher et al., 1998) reviewed these assessments and concluded that use of the risk assessment approach can add significant value to watershed management.

Butcher et al. (1998) described the benefits of applying the integrated watershed risk assessment approach in a review of these five case studies, including the following:

- The risk assessment framework can add significant value to watershed-scale management programs that follow the watershed approach, particularly when addressing problems caused by multiple and nonchemical stressors.
- The watershed approach is expected to benefit from use of the formal and scientifically defensible methods of risk assessment for prioritizing and evaluating risk.
- Although best professional judgment may arrive at the same conclusions as an ERA, the process helps people to carefully examine what led them to their conclusions and document their findings.
- Simpler methods will be required for applying watershed ERA to environmental management on a widespread basis.

In addition to the EPA case studies, many other watershed assessments have been published; several of these are referenced in this report. A comprehensive review of watershed assessment literature would require a much larger report and is beyond the scope of this effort.

2.1. WHO WILL USE THIS REPORT?

This report is principally written for environmental scientists and risk assessors seeking to provide scientific information for watershed management. Watershed managers, coordinators, and others participating in watershed assessment may also find this report useful. The report also provides valuable information to researchers, educators, and students in the fields of aquatic ecology, landscape and seascape ecology, ecological risk assessment, and watershed management. Although the focus of this report is on watershed assessment, many of the principles can be applied to any environmental monitoring and assessment activity intended to inform decisionmakers. Finally, with its examples and descriptions of watershed ERA, this report also serves as an easy-to-understand introduction to ERA.

2.2. WHAT IS WATERSHED ECOLOGICAL RISK ASSESSMENT?

Watershed ERA combines the watershed approach with ERA. Before describing EPA's watershed ERA approach, we need to first describe the watershed approach.

Watershed Approach

The watershed approach is organized around the guiding principles of partnerships, geographic focus, and management based on sound science and data. The watershed approach is a framework for coordinating environmental management that focuses public- and private-sector efforts on addressing the highest priority problems within a hydrologically-defined geographic area (U.S. EPA, 1996). The watershed approach should involve the pertinent levels of government, users of watershed resources, environmental groups, those believed to cause environmental problems, and the public and should help them better understand the specific problems in hand, identify and agree on goals and priorities, and choose and implement solutions. Because watersheds often cross political boundaries, it is important to involve stakeholders across these boundaries.

EPA and other agencies are increasingly using watersheds for environmental management because watersheds are naturally cohesive hydrologic units that are spatially appropriate for management actions (Maxwell, 1998; U.S. EPA, 1995a). For example, the Tennessee Valley Authority has River Action Teams, in which members share monitoring information with key stakeholders (i.e., regulatory agencies, state and local governments, businesses and industries, citizen-based action groups, and watershed residents) to obtain their support in developing and implementing protection and mitigation plans (U.S. EPA, 2002a). The Canaan Valley Task Force in West Virginia produced an inventory of environmental stressors associated with ecosystem problems and their causes, determined whether the problems are getting worse, and developed solutions (U.S. EPA, 1997a). In addition, many other states have implemented watershed restoration actions in response to the Clean Water Act's Section 319, nonpoint source pollution program (U.S. EPA, 2002b). These strategies focus management actions on geographic regions rather than on specific media (e.g., air or water).

The prominent aspects of the watershed approach and comparable aspects of the ERA process are shown in Table 1. Assessment is one of the most critically important parts of watershed management because it attempts to transform scientific data into policy-relevant information that can support decision making and action. A watershed assessment may be initiated by any concerned party in the watershed, including citizen groups, regional coalition members, municipal officers, decisionmakers, litigants, ecological scientists, and a number of other stakeholders regardless of title or association. Watershed ERA provides a basis for estimating ecological effects as a function of exposure to various sources of stress in the watershed. This approach helps environmental managers focus analyses on the highest priority problems in the watershed.

Table 1. Prominent aspects of the watershed approach and comparable aspects of the ecological risk assessment process

Watershed Management	Ecological Risk Assessment
Geographic focus	The scope of the assessment is identified during problem formulation.
Partnerships and stakeholder involvement	Interactions with managers and stakeholders are encouraged, particularly during problem formulation and risk characterization phases.
Continuous improvement based on sound science	The analysis and risk characterization phases provide and organize scientific information relevant to management decisions.

Source: Serveiss (2002).

2.3. THE WATERSHED ERA PROCESS

Watershed ERA integrates the central aspects of the watershed approach, hydrologically defined geographic boundaries, stakeholder involvement, and sound management, with ERA. Although the ERA process is presented in a linear fashion, in reality it is an iterative process that includes a regularly occurring dialogue between scientists and managers. This iterative process is represented by the two-sided "arrows" in Figure 1 and by the vertical box at the right of the diagram denoting tasks that may need to be revisited along the way. Stakeholders should be involved to ensure relevancy to their concerns. The iterations and risk communication take on

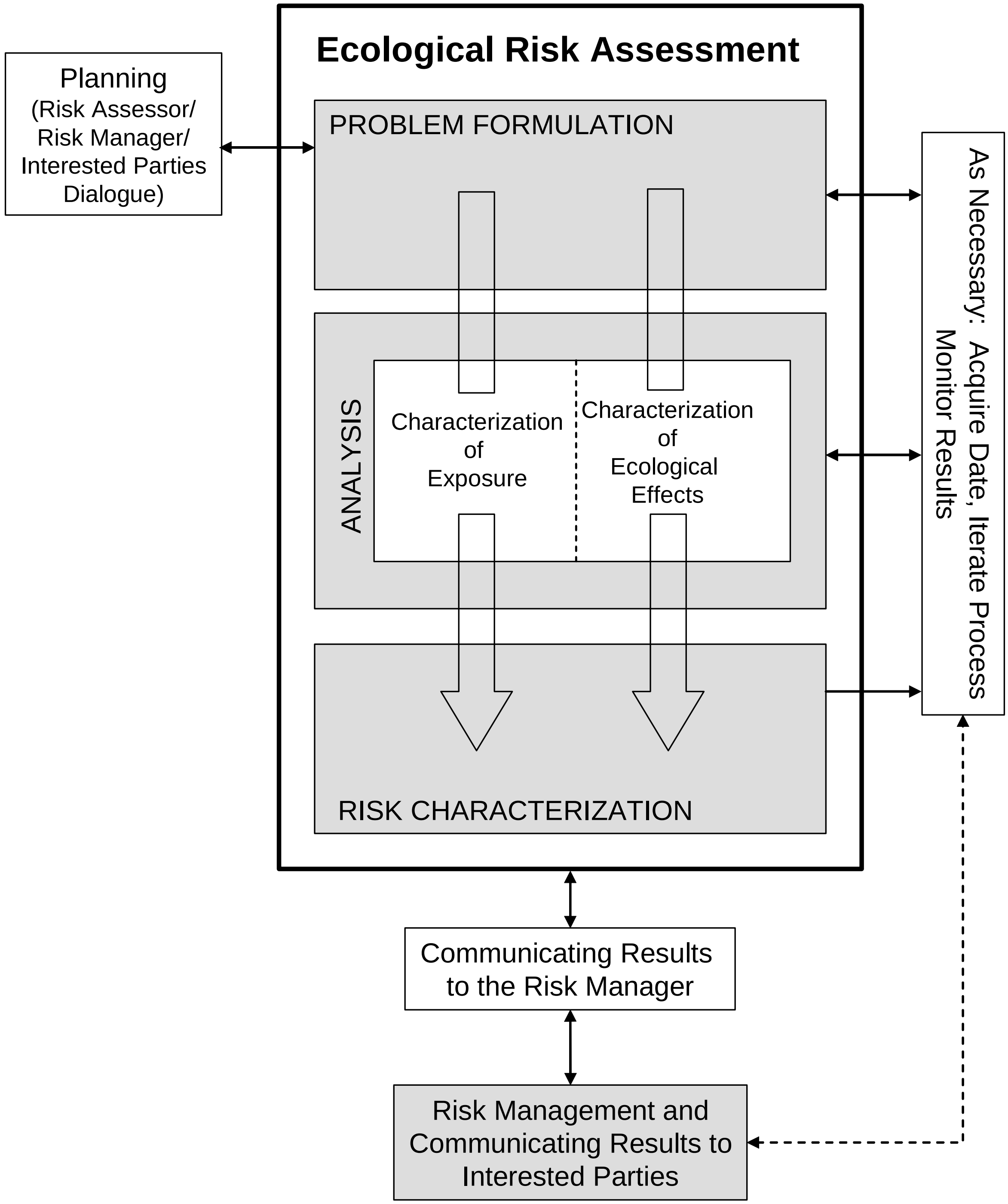

Figure 1. The ecological risk assessment framework.

Source: U.S. EPA (1998a).

greater importance in a watershed assessment with multiple stressors, pathways, ecological resources, and diverse managerial and stakeholder interests (Serveiss, 2002). Based on the nature and timing of risk management needs, scientific findings, and resources available for the assessment, various phases of watershed ERA may need to be revisited repeatedly or skipped entirely. For presentation purposes, this report covers the six steps of ERA sequentially: planning, problem formulation, risk analysis, risk characterization, risk communication, and risk management.

Planning

During planning, the stakeholders are identified and assembled with the goal of sharing information. Their input is needed to establish the focus, scope, and complexity of the risk assessment. The workgroup needs to identify the boundaries for study (see Chapter 3).

Problem Formulation

During problem formulation, assessment objectives are refined and information about the watershed and resources at risk is collected and shared. In addition, assessment endpoints are selected. A conceptual model is created showing pathways between sources, stressors, effects, and assessment endpoints. The associated risk hypotheses are described and environmental management options are developed based on stakeholder input. Finally, an analysis plan is developed to guide future steps of the assessment (see Chapter 4).

Risk Analysis

The objectives of the risk analysis phase are to gain a better understanding of (1) the extent to which ecological resources have been or will be exposed to the most important environmental changes resulting from human activities, and (2) what effects are likely to occur or have already occurred as a result. Characterizations of exposures and effects can take the form of graphs, models, maps, or other illustrations of the relationships among sources, stressors, and measures of effect. These characterizations can be complex because of multiple interactions among the various watershed ecosystem components (see Chapter 5).

Risk Characterization

Risk characterization describes the linkages between exposure and potential effects. The lines of evidence supporting findings need to be presented along with the uncertainty. Ideally, risks are associated with the various management alternatives to support the risk management process (see Chapter 6).

Risk Communication

Risk communication begins with planning, then continues through the assessment, and concludes with the presentation of results. Scientists or assessors share information with managers or stakeholders to ensure that the most fruitful analyses are performed to provide meaningful information for environmental management (see Chapter 7).

Risk Management

Risk management is the decision making step of the process. Attempts are made to select management options that will reduce or minimize risk and achieve objectives that have societal value (see Chapter 8).

3. PLANNING PHASE

To make sure ERA results support decision making, risk assessment participants plan their activities at the start of the assessment process. During planning (Figure 1), scientists, managers, and stakeholders discuss the focus, scope, and complexity of the risk assessment. Participants should determine whether watershed ERA is an appropriate tool to address environmental management concerns because a sound scientific approach is not without cost.

Scientists, managers, and stakeholders all play a role in watershed ERA. Scientists (e.g., risk assessors, hydrologists, and ecologists) need to communicate what they can realistically provide to the managers and the level of uncertainty. Managers must describe why the risk assessment is needed, its relevance to regulations, and what they expect to do with the information they will receive. They must also identify and include the opinions of all appropriate stakeholders, such as ecological and socioeconomic concerns. Managers then need to develop a list of environmental goals and objectives that reflect the stakeholder input. Without this information, assessment results may not be useful to decision making. The success of a watershed ERA strongly depends on the quality of communication that occurs during this initial planning process.

The watershed approach (U.S. EPA, 1995a) defines a similar, although less linear and more problem-oriented, planning process (Figure 2). The watershed approach is contingent on building a project team and obtaining public support. One of the team's initial activities is defining the problem: generating a mutually agreeable problem statement that defines future activities. Specific goals and environmental objectives are then developed to address perceived problems. Goals and objectives are based on the condition or vulnerability of valued resources and beneficial uses, the needs of the ecosystem, and stakeholder's needs (U.S. EPA, 1996).

The key aspects of planning, as outlined in this chapter, include:

- Identifying and including stakeholders
- Setting environmental management goals and objectives
- Defining management options
- Determining the focus, scope, and complexity of the risk assessment

3.1. IDENTIFYING AND INCLUDING STAKEHOLDERS

The planning phase for a watershed assessment is especially complex because a watershed typically overlaps multiple jurisdictions that are managed by organizations with divergent goals and responsibilities and inhabited by numerous stakeholders with varied interests.

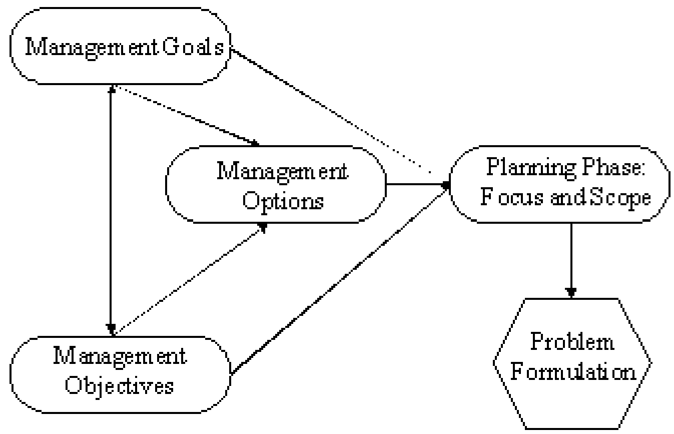

Figure 2. Relationships between management or ecological risk assessment (ERA) goals and objectives, management options, the planning phase, and subsequent problem formulation in Watershed ERA.

Depending on the perspectives of the stakeholders involved, a different set of goals and objectives may emerge. Stakeholder participation may become complex if there are numerous stakeholders or some of them are interested in only a limited range of issues (Glicken, 2000). The inadvertent exclusion of a stakeholder group may influence a group's decision to accept or reject the outcome of the process. Excluded stakeholders could even take legal actions, such as filing a citizen lawsuit under the Clean Water Act to challenge the process outcomes. Therefore, stakeholder involvement needs to be balanced against the limited resources available for watershed assessment and management (U.S. EPA , 2001a).

Participants in the watershed ERA process should include all regulatory or resource agencies (state, federal, local) with responsibilities for protecting and managing the water body and any parties whose authority will be needed to implement a management plan (e.g., local and county officials with zoning oversight). Nongovernmental organizations (such as watershed associations or councils, river watch citizen groups, volunteer monitoring group, educational and research institutions, industries, and agricultural associations) all have a stake in watershed management. Besides organized groups, other stakeholders include landowners, those who use the watershed, and those whose participation is essential to successful management. In some instances, stakeholders may be hundreds of miles away from the assessment (e.g., bird watchers concerned about migratory waterfowl). The nongovernmental and unaffiliated stakeholders may have objectives that are very different from those of the regulatory agencies (e.g., minimizing restrictions on land use, resource development, or waste disposal). Although such social, legal, and economic objectives may be in conflict with some environmental objectives, they are still relevant concerns that need to be considered (Stahl et al., 1999). To help assess these tradeoffs, it may be helpful to involve environmental economists.

Participants on a watershed assessment team contribute different resources that include socioeconomic information, historical data, scientific expertise, and assets to conduct the assessment. Team members of the Big Darby Creek assessment project included biologists, a city planner, and environmental scientists (Cormier et al., 2000). Team members performed literature reviews and frequently consulted or interviewed experts in other disciplines. The team interacted regularly with the Big Darby Partners, a group of state agencies, representatives from The Nature Conservancy (TNC), and farmers concerned about the future of the working landscape.

The various EPA-sponsored watershed assessment teams found that continuity of project contact people, face-to-face contact, and formal and informal group discussions were important to building the trust necessary to engage stakeholders. Continuity with federal or state leaders is also important. A watershed assessment coordinator is generally needed to keep participants focused and on track, and to communicate progress to the stakeholders (U.S. EPA, 1997b).

Stakeholder exclusions may be acceptable if a trusted member of another group adequately represents their opinion. In the Clinch and Powell Valley assessment (U.S. EPA, 2002a), for example, an interdisciplinary workgroup was established with representatives from the U.S. Fish and Wildlife Service (FWS), Tennessee Valley Authority (TVA), TNC, EPA, and other state and federal agencies. Although many other stakeholders were not directly involved in the Clinch and Powell assessment, TNC was able to represent their concerns because TNC interacts regularly with the public and used results from a recently completed survey. EPA provides further guidance on identifying and engaging stakeholders (U.S. EPA, 2001b, 2000c).

3.2. SETTING ENVIRONMENTAL MANAGEMENT GOALS AND OBJECTIVES

Managers must implement decisions to achieve environmental management goals. Selected decisions often begin as one of several management options identified during the planning phase. Clear management goals must be set before the assessment team jumps into analysis. It is quite common that assessments often hastily include a few obvious goals and then devote most of the effort to data gathering and analysis (Reckhow, 1994). Valuable data may be collected, but if the goals are poorly defined, the data may not be useful in management decisions (Ward, 1996).

Before beginning the actual assessment, managers, with input from scientists and stakeholders, need to develop watershed management goals. Some goals are clearly spelled out in legislation and regulation (e.g., support for designated uses under the Clean Water Act). Other times, goals may not be defined, such as "apply an ecosystem approach to management." Elements of existing goal statements from watershed councils, conservation plans, or local growth planning strategies should be incorporated where appropriate. Goals may relate to protecting, restoring or maintaining a species, community, or ecosystem, and they should be explicit and quantifiable.

When goals are broad, it may be useful to break them down into multiple management objectives. Objectives may translate the goals–which may be very general, abstract, and impossible to measure–into specific characteristics that are useful for deciding among management alternatives. The process of defining objectives generally reflects a stakeholder vision of the future condition of the watershed (U.S. EPA, 1995a), and plentiful examples are available from successful watershed projects (U.S. EPA, 1997a).

It may be useful to develop lists of draft objectives representing all stakeholder perspectives using a facilitated interview process with individual members of a stakeholder advisory group (McDaniels, 2000; Gregory and Keeney, 1994). A watershed coordinator (or watershed assessment coordinator) can then condense the lists of draft goals and objectives into a single coherent set. Frequently, stakeholders have differences, and sometimes they have contradictory perspectives. Some goals may be more politically favorable than others. A group could start with general environmental goals (e.g., control eutrophication and protect salmonid fishery) and then make them into specific objectives (e.g., prevent nuisance blue-green algal blooms and provide habitat for successful trout spawning). Reckhow (1994) provides an example of the objectives hierarchy process for Lake Okeechobee. At the top level is the overall goal to "manage the eutrophication of Lake Okeechobee." The next level is subdivided into nine specific goals, ranging from "minimize costs" to "protect threatened and endangered species." These goals are further subdivided by defining specific objectives, such as maintaining the populations of specific wading bird species. As the objectives become more refined, they are likely to naturally point the way to useful measurement endpoints for use in the analysis.

To illustrate the process used in establishing a goal and objectives, Big Darby Partners, composed of managers and a stakeholder group, developed a management goal, "protect and maintain native stream communities of the Big Darby Creek ecosystem" (Cormier et al., 2000). Three objectives for accomplishing that goal were established:

1. Attain water quality criteria for designated uses throughout the watershed.

2. Maintain exceptional warmwater criteria for stream segments that were designated between 1990 and 1995.
3. Ensure the continued existence of native species in the watershed.

These were very useful objectives because Ohio's Water Quality Standards specifically link water quality to the ability of a stream to support and maintain native species. For instance, several streams in the watershed, including the Big Darby and Little Darby, are classified as Exceptional Warmwater Habitat waters. These water bodies have species composition, diversity, and functional organization comparable to the 75th percentile of statewide designated least impacted reference sites.

The Waquoit Bay interdisciplinary and interagency assessment team developed an environmental management goal for the watershed through a multistep planning process (Serveiss et al., 2004). The process included a public meeting to initiate the assessment and understand public values, a meeting of team members to develop the goal and more specific objectives, and a meeting with local resource managers to refine the goal and objectives. The following goal was mutually agreed upon: Reestablish and maintain water quality and habitat conditions in Waquoit Bay and associated wetlands, freshwater rivers, and ponds to (1) support diverse, self-sustaining commercial, recreational, and native fish and shellfish populations and (2) reverse ongoing degradation of ecological resources in the watershed.

The Waquoit Bay assessment team identified 10 objectives that more explicitly stated the kinds of management results implied in the general goal (U.S. EPA, 2002c):

1. Reduce or eliminate hypoxic (low oxygen) level or anoxic (no oxygen) events.
2. Prevent toxic levels of contamination in water, sediments, and biota.
3. Restore and maintain self-sustaining native fish populations and their habitat.
4. Reestablish viable eelgrass meadows and associated aquatic communities in the bay.
5. Reestablish a self-sustaining scallop population that can support a viable fishery.
6. Protect shellfish beds from bacterial contamination that results in bed closures.
7. Reduce or eliminate nuisance macroalgal growth.
8. Prevent eutrophication of rivers and ponds.
9. Maintain diversity of native biotic communities.
10. Maintain diversity of water-dependent wildlife.

Ideally, these objectives would be quantified to evaluate environmental improvements from an established baseline. For instance, objective 1 above could have been quantified as reducing the number of days per year in which dissolved oxygen in any part of Waquoit Bay declined to below 2 parts per million within 5 years. For objective 4, the quantified objective could have been to increase eelgrass cover from 10% to 20% of the bay's water surface area within 5 years.

Although reaching agreement on watershed goals may delay the start of the assessment, reaching agreement among diverse interests is valuable for performing the most useful analyses and for selecting and implementing the most relevant management options.

3.3. DEFINING MANAGEMENT OPTIONS

For assessment findings to be useful, there should be a set of management actions or options that could be implemented pending the results of the assessment. Managers usually must take action to achieve environmental management goals, although no action and passive approaches are also available. A preliminary identification of management options or alternatives should take place as part of the planning phase (U.S. EPA, 1998a).

The Clinch and Powell assessment team identified several management actions that could be implemented to help attain their environmental management goal (Serveiss, 2002), including containing and treating mining runoff, implementing agricultural best management practices, installing roadside spill protection devices, and improving treatment of wastewater discharges. Inclusion of management options early in the ERA process helped them select the spatial scale on which to focus the assessment (i.e., upstream of Norris Dam, TN) and the impacts from mining and agriculture. For more complex watershed assessments, it may be necessary to determine baseline risks as an initial goal, with subsequent iterations evaluating risks relevant to specific management alternatives.

Identification of management alternatives in the planning phase is consistent with the watershed management process, in which the stakeholder objectives and management options together define the context for the management effort. The watershed approach (U.S. EPA, 1995a) further emphasizes this linkage by combining "setting goals and identifying solutions" into a single step in the process.

3.4. DETERMINING THE FOCUS, SCOPE, AND COMPLEXITY OF THE RISK ASSESSMENT

For most natural resource management decisions, ample time and resources to conduct a rigorous assessment of the impacts of multiple stressors are rarely available, thus limiting the potential scope of the assessment. The focus and scope of the assessment should be determined by how sure a manager needs to be to choose an appropriate management action. Some decisions, due to limited resources or urgency, may require only information obtained during problem formulation, while more costly or controversial decisions may require more detailed evaluations. Spatial and temporal boundaries need to be established at the start of the process, and these should be both relevant to stakeholder objectives and practical from an analysis standpoint. The level of effort necessary to reach a decision ideally would be agreed upon early during planning, before conflicts arise over selecting the analyses to be performed. The key to this challenge is obtaining consensus on where (what size watershed), when (what time period), and how much (cost) to study.

Using a tiered approach can help control the level of effort. Each tier would represent increasing levels of complexity and investment and decreasing levels of uncertainty. When tiers are used, specific descriptions of management questions and decision criteria for each tier should be included in the analysis plan (U.S. EPA, 1998a).

Another approach for reducing the level of effort is to concentrate on one portion of the watershed. Only the free-flowing upper portion of the Clinch and Powell Rivers was analyzed because this was believed to be the major refugia for a number of native aquatic species (U.S. EPA, 2002a). It was also decided to focus limited resources on analyzing previously collected data and not to collect additional samples.

3.5. PRODUCTS OF THE PLANNING PHASE

Planning is complete when participants have reached consensus on the following:

- Environmental management goals and objectives for the watershed
- Potential management options to be considered pending assessment results
- Objectives for the risk assessment, including criteria for success
- Focus and scope of the assessment, including the geographic boundary
- Technical and financial resources to be invested in the assessment

Ongoing planning activities that will sustain the assessment include coordination of participants, assignment of administrative or data collection tasks, scheduling regular meetings, and setting due dates for assessment products. Much planning occurs early, but because this is an iterative process, planning activities are often revisited. Management goals, objectives, and options along with the assessment, focus, scope and complexity can all change based on new information.

4. PROBLEM FORMULATION PHASE

Problem formulation develops and evaluates potential causes of ecological effects. This phase provides the organizing framework upon which the entire ERA depends (U.S. EPA, 2000b). Problem formulation begins by reviewing available information and describing and delineating the place and system of interest. Relevant information includes habitat and ecosystem type, general geography, major animal and plant groups, location, boundaries, and dominant land uses. The goal is to assemble information on the potential stressors contributing to either observed or potential alterations in watershed conditions and processes. The assessment will strive to improve the understanding of these relationships, provided resources to do so are available. In some cases, for complex systems such as watersheds, completing just the problem formulation phase may yield an effective standalone management product without proceeding to the risk analysis, risk characterization, risk communication, or risk management phases (Serveiss, 2002).

Problem formulation leads to causal analysis (i.e., what processes could have caused deleterious environmental effects). A comprehensive understanding of the ecological interactions within a watershed is one of the most powerful tools for informed planning, participatory decision making, and logical problem formulation. The paradigms that have been developed to explain the structure and function of stream and river ecosystems serve as examples of the basic knowledge needed by the practitioner. These paradigms include the river continuum concept (Vannote et al., 1980), the flood pulse concept (Junk et al., 1989), influences of the riparian corridor (Stewart et al., 2001), the nutrient spiraling concept (Newbold et al., 1982), the serial discontinuity concept (Ward and Stanford, 1983), and the patch dynamics concept (Townsend, 1989).

The following basic elements of problem formulation are outlined in this chapter:

- Identifying stressors and sources
- Selecting assessment endpoints and other measures that will be needed to quantify risks
- Developing a conceptual model describing predicted relationships among sources, stressors, and assessment endpoints
- Evaluating assessment endpoint
- Creating an analysis plan to guide the next phase of the assessment

4.1. IDENTIFY STRESSORS AND SOURCES

Stressors are defined as any chemical, physical, or biological entity that can cause an adverse effect on an assessment endpoint. Examples of stressors include pesticides, altered stream flow, and invasive species. Typically a wide range of stressors may affect a given resource or assessment endpoint. Stressors may originate from a variety of sources, including different human activities and natural processes.

Watersheds are challenging from an ERA standpoint because they generally have multiple potential stressors and sources of stressors. Sources are actions that release or impose a stressor on the environment. The sources of the stressors are often more manageable than the stressors themselves in watershed ERA and are therefore the focus of management options.

When multiple sources are identified, it may be efficient to focus only on sources that management can control. Nutrients (nitrogen in particular) were identified as the cause of observed algal blooms in Waquoit Bay (U.S. EPA, 2002c), resulting in declines in commercial fisheries. Management options focused on limiting excess nutrients originating from septic systems or from various land uses in the watershed, rather than from atmospheric deposition, because nitrogen inputs within the watershed were found to be more controllable.

Depending on the properties of the watershed and the assessment endpoints, a large number of stressors may be identified. The decision about which pathways and stressors to analyze should be made by using the collective best professional judgment of an interdisciplinary team with expertise encompassing the spectrum of the system under study (Foran and Ferenc, 1999). Foran and Ferenc (1999) summarize a variety of techniques, the vast majority of which involve having team members score the impact of stressors on valued ecological resources. Each individual team member can enter a score, or groups of individuals can arrive at a consensus through discussion. Scores can then be summed or averaged to determine impacts, and a matrix can be devised to determine potential remediation options (Harris et al., 1994).

This scoring approach to stressor prioritization was used in the Waquoit Bay assessment, in which a preliminary list was developed showing the major stressors that included nutrients, toxic chemicals, suspended sediments, and physical habitat alterations (Serveiss et al., 2004). An impact matrix, with stressors as rows and assessment endpoints as columns, is derived from the conceptual model (Table 2). The interdisciplinary workgroup used their best professional judgment to rank the stressors and their perceived impact on assessment endpoints on a scale of 1 to 5. This decision analysis method for ranking alternatives according to multiple criteria is known as fuzzy-set logic (Harris et al., 1994; Wenger and Rong, 1987). Going through this stressor ranking process helped the assessment team justify use of limited resources to analyze impacts on the two stressor-assessment endpoint pathways related to effects of nutrients on scallops and eelgrass. Assessment endpoints are discussed in the next section.

Table 2. Effects matrix summarizing assumed strength of relationships between assessment endpoints identified (e.g., percent eelgrass cover) and stressors in the Waquoit Bay watershed[a]

	Assessment endpoints ranking							
Stressor	**Percent eelgrass**	**Finfish diversity**	**Scallop abundance**	**Anadromous fish**	**Wetland birds**	**Piping plovers**	**Fish/ shellfish**	**Totals**
Chemical pollution	1	1	1	1	1	1	3	**9**
Altered freshwater flow	1	1	1	2	3	1	1	**10**
Nutrient enrichment	5	5	5	3	2	1	1	**22**
Physical alteration of habitat	2	1	2	1	2	3	1	**12**
Fishing pressure	1	1	2	3	1	1	1	**10**
Pathogens	2	1	1	1	1	1	3	**10**
Totals	**12**	**10**	**12**	**11**	**10**	**8**	**10**	**73**

[a]Each cell represents the relative effect of a stressor on an endpoint. The ranking (1 = minor, 5 = severe) reflects experience with the likely effects specifically for the Waquoit Bay watershed.

Source: Adapted from Serveiss et al. (2004).

Stressor Identification Evaluation (SIE) is a logical scientific process that has been developed to evaluate available information, identify relevant stressors, and determine the stressors that are most likely causing observed biological impairments. SIE (U.S. EPA, 2000d) was specifically designed to determine probable causes(s), given that an undesirable effect has already occurred. Drawing from standard ecological risk principles, the SIE process works backward from the observed effect to identify possible pathways (causal linkages) that could cause such an effect, the stressors that would lead to those causal linkages, and the sources that would elicit those stressors. As part of this process, potential relevant stressors in the watershed are identified, which, along with the proposed causal linkages and sources, are used to construct a conceptual model as discussed in Section 4.3 of this report. A recent addition to SIE is the Causal Analysis Diagnosis Decision Information System (CADDIS) which can be accessed at http://www.epa.gov/caddis. CADDIS helps users apply SIEs to streams. CADDIS includes information and many examples of stressor-response relationships that can help ecologists and assessors identify possible stressors in their systems (see Section 5.1).

4.2. SELECT ASSESSMENT ENDPOINTS

In the planning process, participants identified general goals and specific environmental management objectives. Assessment endpoints translate these objectives into something scientifically important and measurable. They also provide direction for future analyses. Assessment endpoints consist of an entity and an attribute. The entity is the species (e.g., eelgrass), functional group (e.g., piscivores), community (e.g., benthic invertebrates), or ecosystem (e.g., lake) that must be protected. The attribute is the characteristic of the entity that is important to protect. For example, for eelgrass it could be acres of coverage, and for a lake it could be frequency and severity of algal blooms.

It is a challenge to select the most useful assessment endpoints. Watersheds have a diverse array of stakeholders who may have competing management objectives. To the extent possible, assessment endpoints should represent the range of objectives of all stakeholders. They should be selected based on three criteria: their relevance to the management objectives, their importance in the ecosystem, and their susceptibility to stressors (U.S. EPA, 1998a).

For example, in the Clinch and Powell assessment, protection of native and endangered mussels and fish was an ecologically relevant management objective. Two assessment endpoints were examined in that ERA: recruitment and reproduction of native mussels; and richness, distribution, and abundance of native fish (Diamond et al., 2002; U.S. EPA, 2002a). Native mussels and fish were assumed to be sensitive to the multiple stressors in the watershed.

It is important to tie the assessment endpoint to the overall management goal to make the results directly applicable for resource management. If the overall goal is protection and restoration of native and endangered mussels, for example, the assessment endpoint might specify a relevant well-described species and an attribute representative of the number of individuals of this species in the area of interest. Such specificity in the assessment endpoint can reveal the types of analyses or models needed to determine causality and risk in the risk analysis phase, and will help diffuse potential controversies that may arise among different stakeholders.

In the Waquoit Bay ERA, all of the assessment endpoints were potentially vulnerable to certain watershed stressors (see Table 2), and they reflect the management objectives and overall goals: (1) support diverse, self-sustaining commercial, recreational, and native fish and shellfish populations; and (2) reverse ongoing degradation of ecological resources in this watershed (Serveiss et al., 2004). However, only two assessment endpoints (eelgrass cover and scallop abundance) were analyzed in detail because of their relationship to the predominant stressor.

4.3. DEVELOP CONCEPTUAL MODEL

Conceptual models describe key causal relationships that will be evaluated in the risk assessment. They include a diagram of the predicted relationships between ecological effects and stressors, and ideally they also include a written description of the relationships. Conceptual models are necessary for watershed and landscape assessments because they describe the multiple physical, chemical, and biological stressors and their sources in a system, as well as the pathways by which they are likely to affect assessment endpoints (Suter, 1999) (see Figure 3). At this point in the watershed ERA, these potential causal relationships are based on professional judgments and knowledge and are not yet quantified.

Sources, stressors, and effects are interconnected; thus, it is a challenge to evaluate their relationships to one another. Furthermore, each pathway and each endpoint may have professional advocates and detractors. Therefore, achieving a focus for the assessment may be difficult. Conceptual models help address this challenge.

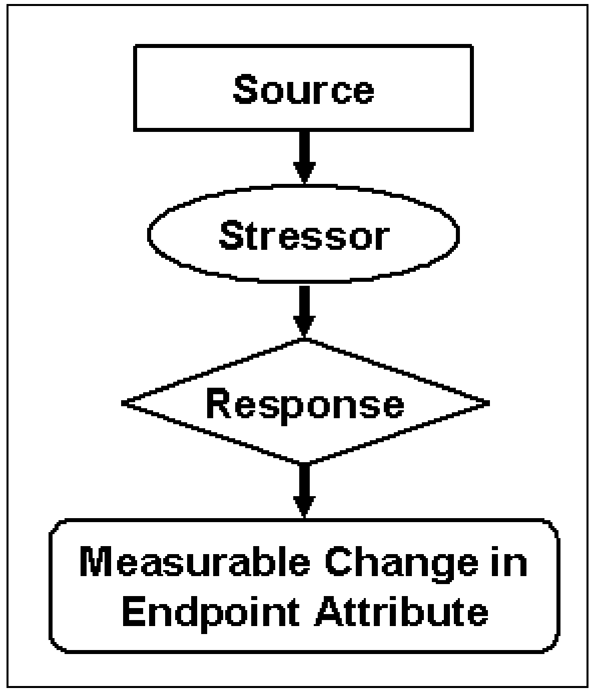

Figure 3. Elementary conceptual model

Developing the conceptual model provides a forum for discussing causal pathways, a framework for explaining the hypothetical relationships and the scope of the assessment, and a structure for the forthcoming analyses. Many specialists may find it easier to see the big picture with the aid of a conceptual model. This systematic process provides a forum and documentation to management actions and helps to efficiently bring new personnel up to speed. U.S. EPA (2005) provides a more in-depth discussion on the applicability of conceptual models

Conceptual model development has been identified as the single most valuable component of EPA's watershed ERA prototype assessments (Serveiss, 2002; Butcher et al., 1998). For instance, in the assessment of the middle segment of the Snake River, the conceptual model and associated risk hypotheses provided a common basis for coordinating the concerns of EPA, FWS, the Federal Energy Regulatory Commission, and the public (U.S. EPA, 2002d). In the Big Darby Creek assessment, group efforts to develop the conceptual model and risk hypothesis were particularly valuable for communicating expectations within the technical workgroup and to the stakeholders.

An interdisciplinary scientific team, including individuals with local ecological knowledge, should develop the conceptual models for the watershed based on previously collected information and their best professional judgment. Early models may be simple, but they provide a basis for predicting relationships between sources, stressors, and effect, and for identifying knowledge gaps. The predicted relationships provide qualitative forecasts of the impact of stressors on resources, and qualitative hypotheses may also be helpful for decision making. These predictions may be used to prioritize problems and help address TMDLs, which are pollution budgets to reduce loadings of pollutants that exceed water quality criteria.

Figure 4 is an example of a complex conceptual model with various pathways organized around stressors. Each stressor has a patterned line that illustrates a pathway connecting its sources to possible effects and selected endpoints. Each model component is represented by a different geometric shape to aid interpretation. This is a broad-based model that provides a framework for the risk assessment and an overview of ecosystem processes. The diagram shows only stressors and effects thought to be potentially important in the Waquoit Bay watershed. The exposure pathways, from the source of stressors to valued resources, are the possible risk hypotheses to be analyzed as part of the risk assessment.

Initial efforts at producing a conceptual model may result in a highly complex set of pathways and linkages. These models are necessarily complex to capture all the pathways between sources, stressors, and effects. It is a challenge to balance complexity with the clarity necessary to understand a diagram with a complicated web of boxes and arrows. One way to achieve this balance is to use hierarchical nested conceptual models (Suter, 1999). Hierarchical nested conceptual models are more detailed and may contain more pathways or information than space permits viewers to see clearly in a general model. Depending on the need, a hierarchical model could be constructed for just one stressor or just one assessment endpoint. These single-stressor and single-endpoint models can be used to provide viewers with a simplified overview along with the capability to examine particular pathways in more detail.

Another option for achieving clarity and complexity is to have the model evolve as more information is developed during an assessment (U.S. EPA, 2005). The initial conceptual model in the Clinch and Powell assessment provided a framework that was used to track the progress of analyses. Areas in need of further data collection or analysis were identified and prioritized. Additional complexity was added over time as the assessment team developed a better understanding of source-stressor-endpoint relationships.

Different forms of communication need to be used for those involved. Scientists will need all the details. For the nontechnical audience, it is better to produce a simplified version of the conceptual model showing only significant linkages. For example, the generalized version of the Waquoit Bay conceptual model (U.S. EPA, 2002c) proved to be a powerful communication tool during the risk assessment phase of the process. Its perceived value to stakeholders is illustrated by the decision to display the conceptual model at the Waquoit Bay National Estuarine Research Reserve Visitor Center. The more complex conceptual model can be held in reserve and used to help answer more specific questions.

Another advantage of developing a conceptual model, such as Figure 4, is that it helps elucidate cascading or secondary effects that otherwise may not be immediately apparent.

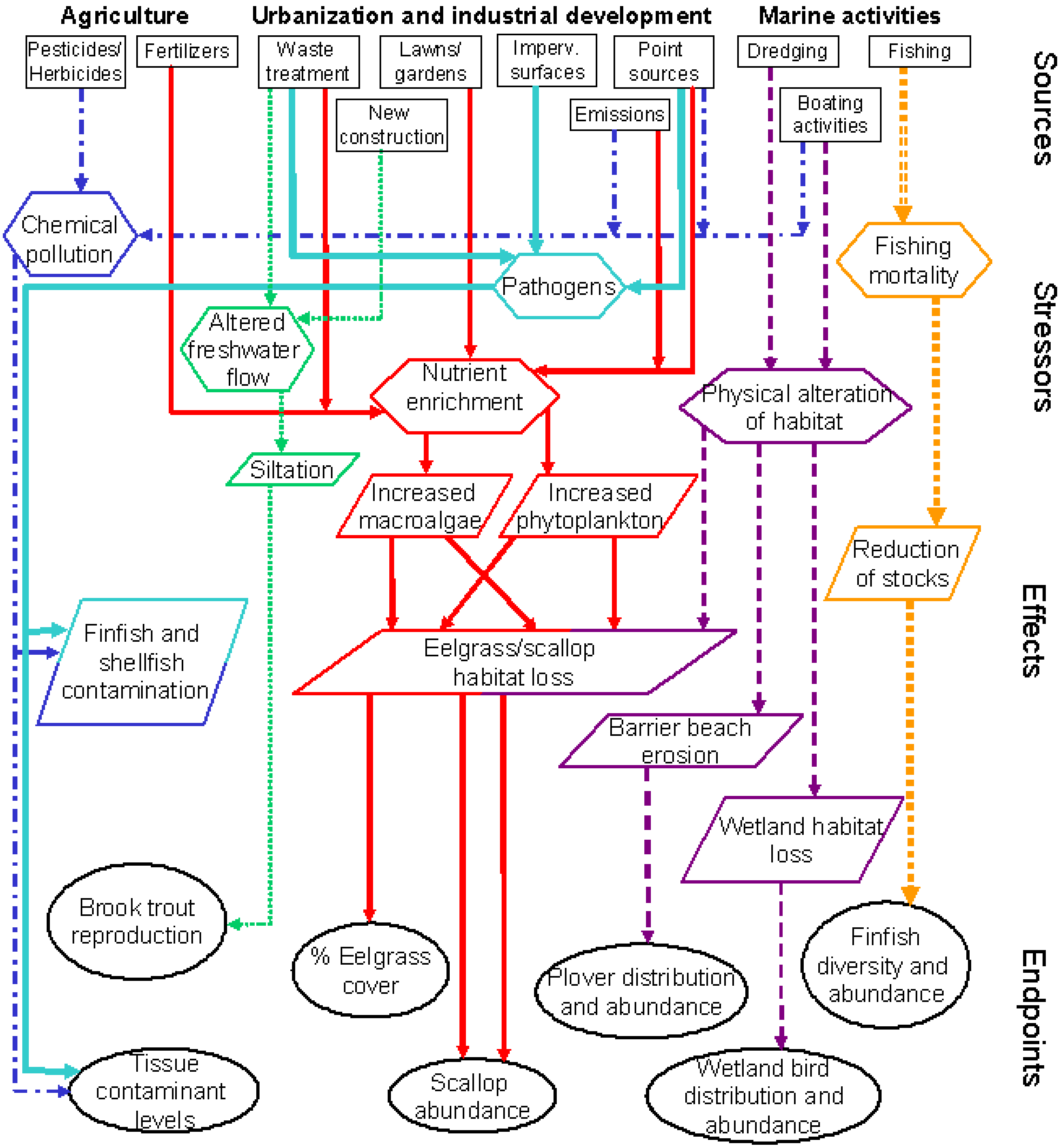

Figure 4. Conceptual model of the Waquoit Bay watershed.
Rectangles represent sources of stressors, hexagons are the specific stressors to the system, trapezoids represent the effects of those stressors, and the ellipses indicate specific endpoints that are affected.

Source: U.S. EPA (2002c).

For example, in Waquoit Bay, excess nutrient input may exert indirect effects on commercial fisheries by causing algal blooms that reduce light levels to the point that some submerged aquatic plants important as habitat for juvenile fish cannot survive (Serveiss et al., 2004). In this case, habitat loss is the stressor impacting the fish, but knowledge of the whole chain of events is necessary to take cost-effective corrective action (U.S. EPA, 2002d).

4.4. EVALUATE ASSESSMENT ENDPOINTS

Assessment endpoints need to be evaluated using observations or modeled predictions. Choosing between management options typically involves a prediction of the response of an assessment endpoint to the proposed management action. However, watershed assessment endpoints themselves are not easily measured, and the response of endpoints to changing conditions is often difficult to predict. For instance, an assessment endpoint of a fishery might be the spawning success of salmonids. This is difficult to measure directly for an entire watershed, although associated information, such as counts of young-of-year from a set of monitoring sites, may be feasible.

Instead of measuring and predicting responses of the assessment endpoint directly, it is often necessary to identify surrogate measures or *measures of effect* that can stand in for the assessment endpoint. Measures of effect are quantifiable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed. Meaningful measures of effect will have the following characteristics (Butcher and Craig, 1998; U.S. EPA, 1998a):

- They will link objectives to stressors (and changes in stressors) and thus correspond to a risk hypothesis defined in the conceptual model.
- They will be sensitive to changes in the condition of the objective.
- They will reflect susceptibility to individual stressors.
- They will be meaningful to decisionmakers.
- They will be measurable, or at a minimum, risk assessors will be able to semi-quantitatively rank them.
- Risk assessors will be able to predict them in response to management options.

Measures of effect are selected for their suitability in reflecting changes likely to occur in the assessment endpoint and for their ability to be measured accurately, consistently, and economically. For example, in the Clinch and Powell assessment, an assessment endpoint was the recruitment and reproduction of native and endangered mussel species. Because data were not available in many parts of the watershed, a surrogate measure of effect was used, mussel species richness. In the Big Darby watershed ERA (Cormier et al., 2000), an assessment endpoint was "biotic integrity of the macroinvertebrate community." The applied measure of effect was the macroinvertebrate Index of Community Integrity (ICI), a quantifiable indices with a standardized protocol that defines the quality of the macroinvertebrate community.

Quantified target values of assessment endpoints should be specified, where applicable, as a means of evaluating and communicating whether a given management alternative meets management objectives. This is particularly important in watershed management, where defined target values can serve as a shorthand to communicate how different management options compare in their ability to meet multiple management objectives. Target values for some measures are already established directly in regulation. For instance, numeric water quality criteria must be met for a management plan to be acceptable, although it may be determined that a more stringent target is appropriate for specific objectives. Where target values are not already defined, a variety of mechanisms can be used to determine an appropriate value. In many cases, the target value will emerge from the ERA itself. Sometimes a narrative standard can be used. It is also possible to set target values by comparing the conditions in the watershed to conditions in an appropriate reference (relatively unimpacted) watershed (U.S. EPA, 1999a), conducting user surveys (e.g., Smeltzer and Heiskary, 1990), or using an existing classification system (e.g., Vollenweider and Kerekes, 1980).

It is desirable to select assessment endpoints and determine whether alternative measures or target values are available before collecting additional data or analyzing them. The next step is to plan the analysis before moving into the risk analysis phase.

4.5. CREATE ANALYSIS PLAN

The analysis plan must describe the objectives of the analysis phase, which are to gain a better understanding of (1) the extent to which assessment endpoints are exposed to important environmental changes from human activities and (2) the effects that are likely to occur as result of exposure. The pathways more likely to impact assessment endpoints or achieve management objectives, along with those more amenable to corrective action, should be investigated. The plan should describe possible results and uncertainties and how this information will be communicated. An integrated and comprehensive data analysis plan also will ensure that all the data needed to define exposure scenarios and pathways to assess ecological risk are clearly identified (Cook et al., 1999). The costs of data collection and analysis have to be weighed against the anticipated risk reduction benefits.

The analysis plan provides a scientifically defensible process to select specific pathways between sources and assessment endpoints for further study in developing meaningful conclusions. The analysis plan should describe the quantifiable (either ordinal or continuous) measures (i.e., both assessment endpoints and stressors/sources) that will be analyzed.

The analysis plan could also include a pilot study intended to test the proposed approach. The Copper Creek subwatershed was studied before the risk assessment team moved on to assess the entire Clinch and Powell watershed (U.S. EPA, 2002a). The analysis plan was modified in response to the pilot study (see Section 5.7 for more details).

When problem formulation is complete, the risk assessment team and stakeholders should have a clear understanding for the assessment and a plan for the analysis phase. Even if the remaining assessment phases are not carried out, problem formulation alone is extremely valuable to watershed management because it summarizes existing relevant ecological information for the watershed, significant data gaps, and potential risks in an organized manner (Serveiss, 2002).

5. RISK ANALYSIS PHASE

The risk analysis phase investigates the most important stressors and exposure pathways and predicts how assessment endpoints respond to stressors under different exposure scenarios. Risk analysis tasks may include collecting data, interpreting historical data, modeling, and statistical analysis. The risk analysis phase ultimately characterizes exposure and effects.

During exposure characterization, the assessor seeks to describe the intensity of the contact or co-occurrence between the stressor and the assessment endpoint (U.S. EPA, 1998a). This involves linking stressors with their sources and then describing where, how, and when the stressors occur in the environment. In traditional ERAs, exposure is commonly estimated by measuring or modeling amounts of stressors and combining these estimates with assumptions about the route of exposure to the assessment endpoint. Spatial and temporal distributions of both the endpoint and the stressors are considered.

During ecological effects characterization, stressor-response data and cause-and-effect relationships are evaluated and extrapolations are made from one set of data to the system being studied. Statistical techniques or mathematical models may be used to quantify and summarize the relationship between stressor and effect. No matter which approach is used, biological monitoring data are very pertinent in identifying the biological consequences of human actions and provide an essential foundation for assessing ecological risks (Karr and Chu, 1997).

The analysis phase of watershed ERA is challenging, because watershed assessments must deal with direct and indirect effects on multiple endpoints from multiple physical, chemical, or biological stressors (Suter, 1999; Hunsaker et al., 1990). The traditional risk assessment paradigm was developed for evaluating risks to single species, or risks from specific chemicals, for which quantitative information is often available or feasible to collect as part of the ERA. For these kinds of single stressor and effect assessments, during the risk analysis phase calculated relationships may be expressed as a stressor-response curve. Although desirable, it may be impractical to develop such a curve in a watershed assessment because of a host of confounding factors, including the presence or absence of other stressors, abiotic influences on the stressor effect (e.g., dissolved carbon and pH affects the toxicity of many metals), and spatial and temporal heterogeneity of the assessment endpoint itself. In many cases, it will be too costly to attain the data for each exposure and effects pathway (Serveiss, 2002). Thus, risk analysis for watershed ecological assessment is likely to require some deviation from the ideal ERA process.

For watershed ERAs, rather than assessing exposure and effects separately, these analyses are usually aggregated. This can lead to merging risk analysis with risk estimation (typically part of the risk characterization phase) because the assessor frequently compares source or stressor data with biological effects data to draw conclusions. As a result, in watershed ERA the distinction between risk analysis and risk characterization can become blurred. Many of the points presented in this chapter on risk analysis could also apply to risk characterization.

Ecological effects are analyzed by describing stressor-response relationships, evaluating evidence for causality, and linking measurable effects to the assessment endpoints identified during problem formulation (Norton et al., 2002a). These components can be developed in any order, and the emphasis may be different depending on whether the objective of the assessment is to predict the effects associated with future change or to retrospectively analyze the causal factors influencing the current state of ecological resources. Tools and challenges associated with watershed risk analysis include the following:

- Identifying stressors and evaluating causes of impairment and associations between causes and effects (Sections 5.1 and 5.2)
- Working with chemical or nonchemical stressors (Sections 5.3 and 5.4)
- Using categorical data or source data or ranking risks because data are limited (Section 5.5)
- Dealing with lots of uncertainty (Section 5.6)
- Recognizing that analysis is an iterative process (Section 5.7)
- Understanding secondary and indirect pathways on valued resources (Section 5.8)

5.1. STRESSOR IDENTIFICATION EVALUATION (SIE) AND CAUSAL EVALUATION: IDENTIFY STRESSORS AND EVALUATE CAUSES OF IMPAIRMENT

SIE is a tool that helps organize scientific information and provides a system to help document the stressors and pathways considered in a risk analysis. The SIE process (U.S. EPA, 2000d), provides a clear and consistent method to identify and compare candidate causes and determine which causes are best supported by the evidence. The process uses a series of three methods. First, candidate causes that are not logically possible given the evidence are eliminated. Then, conventional medical/veterinary diagnostic approaches are used to identify any causal agents that have produced characteristic symptoms. For the remaining causes, a strength-of-evidence analysis is used to evaluate and score all available evidence for each candidate. Based on current experiences in using SIE in several state TMDLs, diagnostic information is often lacking (i.e., data demonstrating a specific stressor or mechanism of effect are often unavailable). Instead, different types of association analysis (temporal and spatial associations between a given stressor and effects), as well as relevant data from the published literature (e.g., tolerance values for a given species and stressor, Habitat Suitability Index [HSI] information, or toxicity endpoints for a given species and chemical) are used to help support or refute hypothesized causal linkages between candidate stressors and observed effects.

Thus, like ERA, SIE promotes risk analyses that are transparent to all stakeholders. In addition, SIE helps ensure that stressors and pathways are not overlooked or prematurely dismissed from consideration, forcing the assessors to consider and evaluate all reasonable sources, stressors, and pathways. SIE has been incorporated into a causal assessment to determine probable causes of biological impairment (Suter et al., 2002), and this methodology has been applied in several places, including the Little Scioto River in Ohio (Norton et al., 2002a, b).

Causal evaluation can identify factors that might be controlled by management actions to improve environmental conditions (Norton et al., 2002a, b; Suter et al., 2002). CADDIS (U.S. EPA, 2004) is a Web-based decision support system that can help scientists in the regions, states, and tribes find, access, use, and share information to determine the causes of biological impairments in aquatic systems. CADDIS is a tool that applies the SIE process to streams and enables scientists to make causal determinations more quickly, cheaply, and defensibly. It helps scientists bring together relevant knowledge on physical, chemical, and biological stressors that may be affecting the aquatic system of concern. This tool is particularly useful for watershed and regional ecological assessments in which scientists must often rely on stressor-effect relationships reported for other sites or in laboratory studies. CADDIS can also help scientists more easily identify relevant data from scientific studies conducted elsewhere and help them organize and compare the evidence across all the candidate causes. Finally, some scientists must be able to clearly communicate the logic of their causal conclusions.

5.2. EVALUATE STRENGTH OF ASSOCIATION BETWEEN CAUSES AND EFFECTS

Both SIE and watershed ERAs evaluate the strength of associations between a particular source or stressor and the effects of concern. The strength of the association (typically evaluated using multivariate analysis) between the stressors (or sources) and measures of effect may indicate cause-and-effect relationships, especially if supported by other lines of evidence (Serveiss, 2002; U.S. EPA, 2000b). In most retrospective watershed ERAs, causes of effects are often inferred from spatial relationships derived using relatively current information. For example, in the Clinch Valley ERA, a strong negative relationship existed between coal mining activities (particularly coal preparation plant discharges) and both fish IBI and number of native mussel species. The particular stressors in this relationship were believed to be both pollutants and sedimentation from coal fines, based on published research in the watershed. The analyses did not suggest that every coal mining activity in the watershed would cause a certain effect on native species, but they did suggest a relatively high probability of risk to native species living in close proximity to coal mining activities. Landis et al. (2000), using a relative risk model (RRM), also inferred cause-effect relationships by associating sources with important habitats in various subareas of the Willamette River Basin in Oregon. This model of analysis yields a ranking of stressors and habitats based on the relative likelihood of occurrence, not on the relative consequence (or effect) of occurrence (Landis et al., 2000).

The Big Darby Creek assessment sought associations between stressors and impacts (Cormier et al., 2000) by relying on current and past land use practices and biological measurements taken at specific sites. Researchers used the ICI for macroinvertebrates and the fish IBI to represent ecological status within stream segments in the watershed. Multivariate analyses were used to determine relationships between index results, instream stressors, and land use patterns in the watershed. The analysis identified community components that were associated with specific types of stress. For example, the percent of *Tanytarsini* midges and *Glypotendipes* increased at sites with low and high biological oxygen demand, respectively. Also, the percentage of darters increased at sites with high scores for stream corridor structure and low concentrations of inorganic nutrients.

5.3. WORK WITH CHEMICAL STRESSORS

Exposure and effects analysis that relate to multiple chemical stressors in a watershed ERA can be handled in much the same way as in single-chemical ERAs. Both types of ERA use nationally developed chemical criteria (e.g., U.S. EPA, 1995b), if available, or effects thresholds obtained from laboratory or field studies are used. A challenge in many watershed ERAs is the likelihood of having multiple chemicals present simultaneously, any or all of which may have caused, or could cause, effects on valued resources. Laboratory-based chemical criteria or thresholds are generally developed independently of other chemicals, or any other stressors in percent, and therefore do not necessarily take into account synergistic or additive effects of multiple chemicals. Some field-based chemical thresholds, such as sediment quality thresholds (Field et al., 2002; Long et al., 1998), may incorporate chemical interactions in their threshold values. Several different tools have been developed to help address the issue of multiple chemical contaminants, the most common of which is to express each chemical concentration as a proportion of its criterion or threshold concentration. These proportions are then added; i.e., the assessor assumes that the effects of the different chemicals are additive. If the sum of the proportions is >1.0, some assessors have interpreted this as a potentially toxic condition. Limited research suggests that additivity may be a reasonable assumption in many cases, but a fair degree of uncertainty is associated with the results of this approach.

One major uncertainty is the concentration of each chemical that is being compared to its respective criterion or threshold. Ideally, one has sufficient data to calculate frequency, duration, and magnitude of concentrations exceeding a given threshold so that the entire record of measurements is used to calculate risk. However, such comprehensive data are rarely available, and the assessor often needs to estimate an average or maximum value with which to analyze risk for the suite of toxic chemicals that might be present.

Another source of uncertainty in analyzing risk of multiple chemicals is the often large variability in sensitivity to the chemicals for different species of interest. While the measures of effect selected to represent the assessment endpoint may help to focus this issue in risk analysis, chemical criteria and thresholds are typically based on many species, and the sensitivity of the types of species that occur in the watershed may be fundamentally different. One tool that has been beneficial, particularly in ERAs involving chemicals for which there are criteria, is a species sensitivity distribution approach (Bossuyt et al., 2005; Newman et al., 2000). This approach examines the sensitivity of all species for which relevant effects data exist and evaluates the distribution of species sensitivity to help predict the portion of the distribution of species that is likely to be affected by a given chemical concentration. For multiple chemicals, such species distributions of risk can be overlaid to obtain an estimate of risk due to chemicals to the aquatic assemblage. The species distribution can also be tailored in some cases to a given region by including only those species, or those representative species, that could reasonably occur in the watershed. In practice, however, this approach is often limited due to a paucity of relevant toxicological data for species of interest. In addition, certain types of species, such as freshwater filter feeders (e.g., unionid mussels) are currently under-represented in toxicological databases because reliable test methods were unavailable. Thus, the database of species sensitivity values available for certain chemicals may be limited and may not include many species or genera of concern in a given watershed.

5.4. WORK WITH NONCHEMICAL STRESSORS

Another challenge regarding stressors in risk analysis, particularly for watersheds, is the fact that many common stressors are nonchemical (e.g., sediment/turbidity, habitat), or are chemicals in media for which readily available criteria or thresholds may not exist (e.g., pesticides in sediment), or are chemicals that are naturally available and necessary for life (e.g., nutrients). Thresholds for these types of pollutants typically require some type of reference condition approach. A reference approach has been commonly used to set thresholds or criteria for biological assemblage condition (e.g., U.S. EPA, 1999a; Karr and Chu, 1997) and is now being extended to other types of parameters that are strongly influenced by natural regional conditions, such as nutrient criteria (U.S. EPA, 2000e). The reference condition approach relies on having measures for the parameter of interest from multiple sites in the same ecological region or ecoregion (Omernick, 1987) and that are minimally or least affected by human-caused activities. Ideally these sites are located in the same watershed or basin as the one being investigated. Thus, these measures ideally reflect natural variability, without the stressor of interest present.

The threshold developed using reference conditions is a statistical attribute of the measures obtained from the various reference sites. A variety of statistical parameters have been used to formulate a threshold. For example, EPA's Rapid Bioassessment Protocols (U.S. EPA, 1999a) recommend, and many states use, the 25th percentile IBI value from the distribution of reference site IBIs as the threshold for reference biological condition. IBIs from a site that are significantly lower than this threshold suggest an undesirable biological condition (i.e., an effect). Clearly, this approach is sensitive to proper selection of reference sites. If more truly impaired sites are identified as reference sites, the established threshold will be less sensitive, and the associated risk analysis will be more uncertain. Also, this referenced threshold approach works best when there are many appropriate reference sites with which to measure natural variability. Biased estimates of natural variability are more likely as fewer reference sites are available. Finding reference sites is a common challenge in certain regions of the United States (e.g., agricultural areas of the Midwest and highly urbanized areas in southern California or the mid-Atlantic coast).

As described in Chapter 3, biocriteria thresholds used in the Big Darby watershed ERA were based on ecoregional reference conditions (Cormier et al., 2000). EPA's nutrient criteria guidance for streams and rivers suggests using a reference approach for developing nutrient criteria (U.S. EPA, 2000f). The EPA report recommends using either the 75th percentile of concentrations observed in ecoregional reference sites or the 25th percentile of measurements obtained from all sites in an ecoregion as a concentration threshold for a given nutrient.

In some ecoregions and watersheds, reasonable reference conditions may be difficult to identify as a result of large-scale human-caused changes to the landscape, hydrology, and aquatic habitats. In these cases, appropriate stressor thresholds may need to be developed using other tools, such as modeled reference conditions, based on multivariate analyses of existing stressor and biological condition data. For example, using quantile regression (Cade et al., 1999; Koenker and Bassett, 1978) of macroinvertebrate Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa richness (a metric that includes many of the pollution-sensitive macroinvertebrate species) and associated urban intensity (a combination of urban land cover and population and road density), resource agencies in southern California were able to identify best attainable biological conditions despite the occurrence of high urbanization and associated stressors. Such analyses can help elucidate causes of biological impairments and provide the means to identify working "thresholds" for a given stressor.

Risk analysis can often take the form of simulation or numerical modeling, depending on the stressors of concern, availability of appropriate criteria or thresholds, and extent of stressor data available. For the middle segment of the Snake River in Idaho, a life cycle analysis was conducted for three highly valued coldwater fish (rainbow trout, mountain whitefish, and white sturgeon), aquatic macrophytes, and endemic molluscs (U.S. EPA, 2002d). These representative species from three major trophic levels (fish, invertebrates, and plants) were chosen as assessment endpoints for the analysis. A simulation model was used to develop quantitative risk estimates of the likelihood of exceeding water quality criteria, macrophyte biomass, and habitat suitability indices for the fish species. The system dynamics were simulated using information on meteorological conditions, hydrological and hydraulic conditions, biological oxygen demand, dissolved oxygen, phytoplankton biomass, various forms of nitrogen and phosphorus, temperature, coliform bacteria, and water depth, all obtained from several locations in the river. Risks to fish and macroinvertebrates were estimated by determining the likelihood of being above or below coldwater biota tolerance limits. Tolerance limits are generally the natural level to which most native species have adapted. In addition, qualitative analyses of exposure and effect were completed using best professional judgment and comparisons with field studies under similar conditions. The results of these individual analyses were combined to draw conclusions in the risk characterization phase.

5.5. WORK WITH DATA LIMITATIONS

5.5.1. Use Categorical Data

All of the EPA-sponsored case studies, and most other large-scale ERAs that have been reported, had to work with some types of data limitations, and therefore uncertainties in their risk analyses. Given the multiple stressors in watersheds, and the relatively large spatial scale of watersheds, most analytical approaches will require fairly large sample sizes to account for the many different potential sources of variation. As a result, data requirements can be enormous for developing meaningful associations between sources, stressors, and effects in watersheds.

While regression-based relationships between a stressor and effect or a source and a stressor are desirable in any ERA, watershed analyses often lack sufficient and appropriate data with which to derive meaningful “stressor-response” curves. One alternative that has been used in many watershed ERAs is to categorize the data before conducting analyses. Categorical data may depict relationships more effectively than continuous (gradient) data in some cases. Both stressor and effect data may be categorized. For example, in the Clinch and Powell ERA, few if any gradient relationships were evident between sources, stressors, and effects. However, relationships became more evident when the measure of effect (i.e., fish IBI), was categorized as impaired or unimpaired (based on ecoregional reference conditions) rather than using the IBI values themselves (Diamond et al., 2002). Similarly, relationships between habitat quality factors, such as sedimentation, and effects were more apparent if habitat quality (the stressor) was categorized as "satisfactory" or "unsatisfactory," based on the resource agency rating system. Categorizing data for a factor, as opposed to relying on the individual data values themselves, often results in increasing the power of the analysis; i.e., there is a higher probability of seeing a relationship if one is there. However, stressor-response relationships may or may not be feasible using categorical data. If only two or three categories of a source, stressor, or effect are appropriate, analyses will often take the form of an analysis of variance (e.g., ANOVA, MANOVA, ANCOVA) or analogous nonparametric analysis to determine statistical differences between categories.

Categorizing data works best when either criteria or consensus-based thresholds for the parameter are available. For example, if the bioassessment data available are based on an index that discriminates four or five categories of condition such as the fish IBI , then it is appropriate to categorize the bioassessment data accordingly for analysis. Analysis of more categories in this case would not be appropriate and could yield misleading results because the index was not calibrated to distinguish between a greater number of categories. Categorizing stressor or source data can be more uncertain because consensus thresholds are typically unavailable. For example, effects on assessment endpoints as a function of distance from a potential source (a surrogate for stressor intensity) is often a useful analysis in watershed ERAs. However, distance from a source may not lend itself to obvious categories unless much is known about the fate and transport of the stressors assumed to be present. In this case, it may be useful to examine the cumulative distribution of the distance of each relevant biological sampling location to its nearest source type of concern (e.g., distance from the nearest active coal mine was examined in the Clinch Valley ERA). Based on this distribution of distances, the assessor can then identify a few categories (e.g., <1 km, 1-5 km, and >5 km), for which a sufficient number of sites exist and can be statistically analyzed in a meaningful way. Again, care should be taken to ensure that the levels or categories are appropriate for the measure being analyzed and the particular watershed.

5.5.2. Use Source Data

Another common data limitation in watershed ERA is that stressor data may not be available or in a useful form. To deal with this limitation, risk assessors have often relied on source data as a substitute. Source data include such parameters as the number of a certain type of point source or the percentage of a particular type of land use. These data are typically fairly easy to obtain for a watershed, and they are readily mapped for analysis and presentation purposes. Source-effect analyses were used both for the Big Darby Creek and the Clinch Valley assessments where land use (source) data were compared with fish and macroinvertebrate (effects) data. Originally, the assessment team for the Clinch and Powell Valley assessment sought associations between fish or mussel data, and stressor data such as sedimentation or water chemistry data, yet data were too sparse for quantitative analyses (U.S. EPA, 2002a). Instead, emphasis was focused more on probable stressor sources and their effects on resources (Diamond et al., 2002) (see Figure 5).

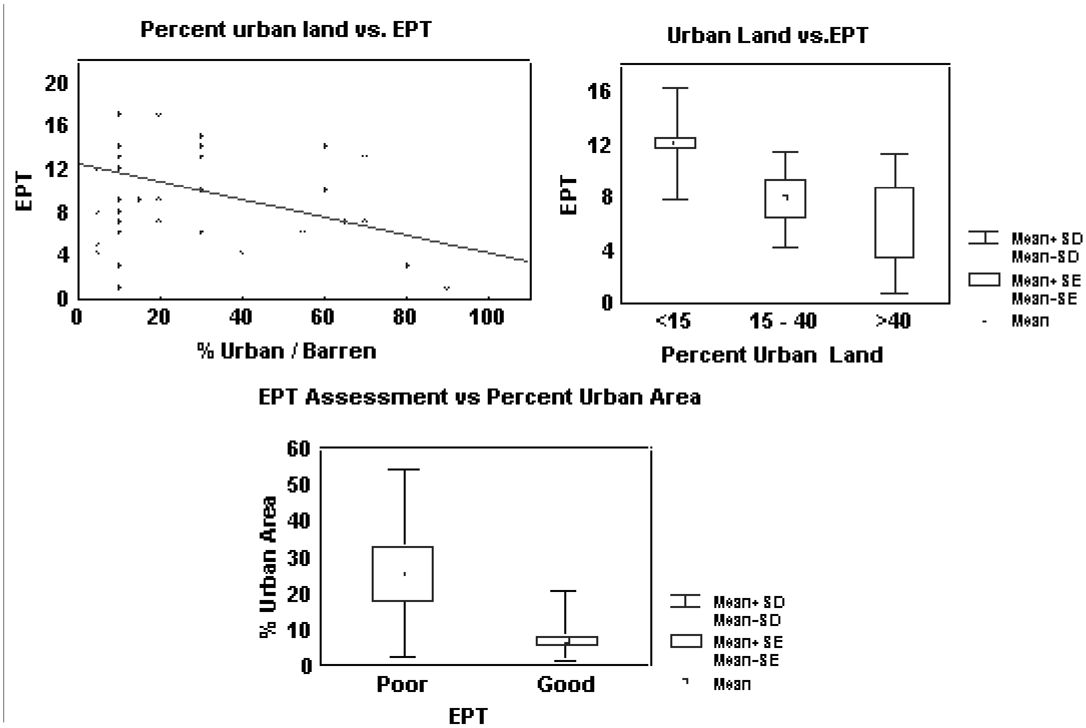

Figure 5. Significant relationships between urban land use activities and invertebrate ephemeroptera, plecoptera, and trichoptera (EPT) score. Relationships between the biological measure (EPT in this case) and land use cover were often difficult to evaluate using linear stressor-response analyses (upper left graph). Relationships were more apparent when examined on a categorical or threshold basis (upper right and lower graphs).

Source: U.S. EPA (2002c).

By comparing a group of sources or stressors to a group of effects, potential exposure of biota to sources of stress can be examined in a variety of ways, depending on the types of data and tools available and the types of sources being analyzed. As watershed ERAs are necessarily place based, spatial analysis tools and GIS are essential to watershed assessment work. Land use (e.g., agricultural and urban) constitutes a major sources of stress identified in watersheds, and these data can be estimated using digital satellite images (e.g., Landsat) and GIS. Typically, percent area cover in a watershed or other defined spatial area is used to quantify the "exposure" of a given land use (e.g., urban development). The way in which land use percentages are calculated may need to be considered carefully. The Clinch and Powell ERA demonstrated that land uses within the riparian corridor were more indicative of exposure to aquatic life than land uses calculated for the watershed as a whole (Diamond et al., 2002). Others have shown similar significance of riparian land uses in watershed studies (Stewart et al., 2001; Hunsaker et al., 1990).

Exposure stemming from different land uses may be inferred from available data in the watershed and exposure-effects information from the literature. For instance, in the Clinch and Powell assessment, the fish community was consistently poor when the surrounding riparian zone included all four main sources of stress: mining, urbanization, major roads, and pasture areas (Diamond and Serveiss, 2001). The strong association of adverse effects on fish in the presence of these nearby land uses was meaningful, even in the absence of more specific, quantitative profiles of exposure and effects (Diamond et al., 2002). However, assessing relationships between land use and ecological resources is especially complex in a watershed, and interactions among land uses could mask relationships between a particular land use and ecological effects. The effects of pasture land could be masked by coal mining effects (Diamond and Serveiss, 2001). In addition, land cover estimates have a certain amount of error associated with them, both spatially and thematically. Thus, depending on when effects or stressor characterization data were collected, land coverages may not be an accurate reflection of source information. These uncertainties need to be recognized and discussed either during this risk analysis phase or during the next phase, risk characterization (Chapter 6). Land coverage data may require "ground-truthing" to ensure that risk assessors are dealing with correct estimates of specific land coverages in a given location or region.

5.5.3. Rank Risks

Occasionally, insufficient data are available to analyze an assessment endpoint or to assess a surrogate measure of effect. In these instances, less precise measures (scalar, ranked, or qualitative) could be used. Relative ranking of risks associated with management options may be sufficiently robust to provide meaningful information to decision makers. The RRM (Landis and

Wiegers, 1997) is one approach that produces a "quantitative" assessment using best professional judgment of exposure and effect rankings.

While the focus of this report primarily has been on EPA-sponsored case studies, a number of regional ecological risk assessments have been developed using the RRM (Walker et al., 2001; Landis et al., 2000; Wiegers et al., 1998). Rather than looking at the traditional risk assessment components of stressor, receptor, and response, the relative risk regional approach examines relationships between sources (groups of stressors), habitat (groups of receptors), and impacts (groups of responses). Stressors are ranked in terms of their occurrence and importance. High relative risk occurs where a source, an impact, and a habitat strongly overlap; where there is little overlap, the relative risk is low. Relative risk can also be summarized via numeric indices, as shown in Wiegers et al. (1998) for the Fjord of Port Valdez. This approach has the clear advantage of prioritizing what are thought to be the main stressors and major habitat types at risk. This analysis can then focus subsequent data collection and analytical efforts on managers concerns.

With the RRM approach, a source can serve as a group of stressors, a habitat can serve as a location for a set of organisms or receptors, and an ecological impact can serve as a group of receptor responses. Each source and habitat is ranked for each area to indicate high, moderate, low, or no risk within the context of the study site. Ranks are assigned using site-specific criteria. RRM ranks sources or stressors in different locations in a region, evaluates or filters if the habitat is affected by sources or stressor, and ranks the impacts. The region is divided into several similar areas based on land use, basins, clusters of similar stressors, habitats, or land uses. Ranks or scores to represent high, medium, low, and none are used to reflect the magnitude of the source and the amount of habitat that could be affected in each subarea. The ranks assigned for each combination of source, habitat, and subarea are multiplied together. If the interaction between the exposure and effect is likely to occur, the two scores are multiplied by one another. The results for each subarea are summed to determine the relative ranking of risk within the region. Thus, by quantitatively determining the relative risks in subregions, the contribution of risks from sources or stressors, risks in various habitats, and risks to assessment endpoints can be calculated. The results can then be portrayed easily in graphical form with maps depicting the relative risks within the study region.

This approach is simple to use and can include stressors about which little is known quantitatively. This approach uses a qualitative, graphical approach for sorting out the many risk pathways that may be present in a regional-scale ERA.

5.6. DEAL WITH UNCERTAINTY

Another source of uncertainty in using source data as a surrogate for stressors is that some land uses or habitat features may be correlated to natural physiographic factors that may themselves have a pronounced effect on biota. In the Clinch Valley ERA, for example, elevation was related to the number of aquatic species present, even in the absence of stressors. Thus, elevation effects had to be “factored out” of the analysis to examine source-effect relationships. This made it necessary for risk assessors to use a smaller subset of the available sites that were within a relatively narrow (i.e., nonsignificant) elevation range for analyses. Further partitioning to account for other factors, such as land uses, would further reduce the number of appropriate sites available for analysis. Depending on the stakeholder concerns, predicted results, and acceptable levels of uncertainty, data may be insufficient for any type of statistical analysis.

The inclusion of natural variability in risk analyses is especially critical at the watershed or larger spatial scales. Aquatic life habitats, for example, are known to vary in quality and quantity depending on natural factors, such as precipitation patterns and geology. At larger spatial scales, these natural factors (including the elevation example described above) are likely to vary across the watershed, necessitating proper assessment of these factors. Typically, natural variability is addressed through the identification of appropriate reference conditions as discussed previously. Measurements of the assessment endpoint, as well as natural factors that affect that endpoint, across several reference sites under a given set of conditions (e.g., precipitation range, elevation), will yield an estimate of natural variability. This variability becomes the basis for determining whether a given source or stressor tends to be associated with a non-natural (i.e., impaired) effect. In some regions, biological data collected over more than a year or season are needed at reference sites to adequately characterize natural variability. This is particularly the case for more arid regions where “wet” years can yield a different biological condition than “dry” years. Because habitat features are often sensitive to natural regional factors, it is important to define natural variability for the watershed in which one is working. The assessor should critically evaluate reference condition data obtained elsewhere to be confident that the information is appropriate for the watershed being evaluated.

An additional uncertainty associated with using source data as a surrogate of stressor information is that all members of a given source class may not have an equal probability of being associated with a hypothesized stressor (or condition). For example, certain agricultural land uses may be associated with sedimentation and excess nutrient input into streams. However, depending on particular BMPs that may have been implemented at a given site, the intensity of these stressors (and therefore predicted ecological effects) may not differ much from that observed under reference conditions (where few or no stressors are present). If such differences among sites are not identified and taken into account in risk analysis, results will be "noisy," masking any real relationships that might exist between certain source types, stressors, and effects. A solution to this challenge is to involve people in the ERA who are knowledgeable about the watershed and can help "ground-truth" risk analyses.

Literature reviews often play an important role because site-specific information about ecological effects, or interpretive benchmarks (e.g., effect or impairment thresholds), is often unavailable for the watershed under study. For example, researchers elsewhere may have demonstrated relationships between the ecological measure of interest (e.g., the abundance and diversity of certain wetland plants) and certain stressors (e.g., urban stormwater runoff) that also occur in the watershed of concern. It may be appropriate to extrapolate results obtained in those other locations to the subject watershed as long as uncertainties are adequately documented.

5.7. RECOGNIZE THAT ANALYSIS IS AN ITERATIVE PROCESS

Another issue of importance is that the collection and analysis of data in the analysis phase may be iterative. Analyses may bring new information to light (e.g., one pathway between stressor and effect may have much greater impact than originally predicted). In addition, management's needs may change or the environmental situation may become dramatically different (e.g., following an oil spill). Many recent recommendations have emphasized that stakeholder and manager involvement needs to be initiated in the planning step, and recurring rounds of deliberations and analysis are necessary throughout the process to make the findings most useful (U.S. EPA, 2000a; Foran and Ferenc, 1999; PCCRARM, 1997; NRC, 1996). New information obtained through literature review, field data, peer review, or environmental changes since the beginning of the assessment may trigger these iterative loops.

As the analysis proceeds, the assessors should present interim findings to managers to ensure that the assessment is targeting the appropriate problems. A lesson learned from the prototype assessments is that regular discussions between the assessors, managers, and stakeholders during the analysis phase are very useful for refocusing the ERA and for reprioritizing pathways for evaluation (Serveiss, 2002). The Clinch Valley ERA demonstrates two particular concepts related to the benefit of dialogue among participants during analysis. The initial risk analysis showed that certain measures were not available or were spatially distributed in a way that made some of the proposed analyses infeasible. Active dialogue among assessors, managers, and stakeholders during this process resulted in modifying the analysis plan and switching to determining associations between land use and biotic measures.

The value of pilot studies also illuminates how watershed ERA is an iterative process. Given the often complex interactions in watersheds, data limitations, and costs, it also may be useful to perform a pilot analysis first to determine that the analytical approach is likely to be completed successfully. Analyzing a subwatershed is one approach for pilot testing the analysis plan (Diamond et al., 2002; MacDonald, 1994). In the Clinch and Powell Valley assessment, the Copper Creek subwatershed was examined first to assess how to apply the analysis plan to the entire watershed (U.S. EPA, 2002a). Copper Creek was chosen for this pilot analysis because it was the most data-rich subwatershed and because it was a relatively simple case in that agricultural uses were the major source of anthropogenic activity. The pilot study was needed to address two analysis objectives central to this assessment: (1) to identify the appropriate spatial scale to test relationships between land use activities or stressors and measures of effect and (2) to identify whether the benthic macroinvertebrate measure (i.e., the EPT index) or the fish IBI would be a reliable surrogate measure of effect for predicting the status of native mussel assemblages. Achieving the latter objective was especially desirable because it was known at the outset of this study that available native mussel data were more limited than either EPT or IBI values.

5.8. UNDERSTAND SECONDARY OR INDIRECT PATHWAYS

In watershed ERAs, more than one source of a given stressor may lead to secondary pathways. For example, toxic chemicals can originate from nonpoint sources such as agriculture (pesticides), point sources (wastewater treatment plants), urban runoff (oils, metals), and many other human activities. The conceptual model should identify the major sources of a given stressor to reveal all pathways. Some stressors, such as temperature, acid runoff, and low dissolved oxygen, may have a seasonal component or may be associated with certain geological or hydrological characteristics of the watershed. The analysis phase should attempt to address such variation if possible to better reflect stressor-response associations.

Secondary stressors may be even a greater concern than primary stressors. Some stressors may have direct results that impair valued resources, such as nutrients resulting in nuisance algal blooms. Nutrients may have secondary effects, however, such as decreased dissolved oxygen, due to increased bacterial respiration and algal die-off. In this case, a secondary stressor from one source can be similar to a primary stressor from another source; e.g., decreased dissolved oxygen may also be a primary stressor resulting from release of oxygen-demanding material (biological oxygen demand) from wastewater effluents or agricultural runoff. These secondary stressors may need to be considered as well. For instance, in Waquoit Bay, nutrient input is a stressor that caused excess macroalgal growth (a secondary stressor) to shade eelgrass and inhibit its growth (Figure 4). Fate and transport models were then used to predict environmental nitrogen loading from the watershed into the bay (U.S. EPA, 2002c). In a similar manner, a model of phosphorus loading in Wister Lake in Oklahoma allowed estimation of changes in lake trophic conditions (Hession et al., 1996).

Risk analysis is complete when quantitative stressor-response characterizations have been completed and assumptions, uncertainty, and data limitations have been discussed. A few iterations may be required to progress from start to finish. Finally, since watershed ERA combines risk analysis with risk characterizations, the necessary iterations may become blurred.

6. RISK CHARACTERIZATION PHASE

The exposure and effects data generated in risk analysis are integrated to form conclusions in the risk characterization phase. Risk characterization should use multiple lines of evidence from models, field work, and statistical extrapolations to increase the confidence in conclusions (U.S. EPA,1998b) and to help provide answers for risk management (see Chapter 8). Finally, uncertainties originating in both the problem formulation and analysis phases should be described (Suter, 1998). The final product of this phase is the risk assessment report prepared for managers to support defensible management decisions.

Risk characterization includes two major steps: risk estimation and risk description. Risk estimation integrates the exposure data and stressor-response estimates from the analysis phase while addressing uncertainties that arose throughout the assessment. Methods used to integrate exposure and effects data include comparing single values of effect and exposure, comparing statistical distributions of exposure and effect values, or conducting simulation modeling. As previously presented, in watershed ERA different techniques may be used to accomplish the integration of exposure and effects data (e.g., causal analysis, risk ranking, using source data) and risk analysis may merge with risk estimation. As a result, risk estimation was discussed in the preceding chapter, and risk description is the focus of this chapter.

6.1. RISK DESCRIPTION

During risk description, risk assessors prepare a summary of ecological risk and interpret ecological significance. Summarizing risk involves making a bottom-line estimate of risk. Ideally, a risk summary would be in the form of a quantitative statement: for example, "There is an 80% chance of a 50% die-off of certain valued fish species in the watershed due to sedimentation from nonpoint sources under current conditions." Such a statement, when combined with information on the reliability of the statement (i.e., the uncertainty), would allow analysis of the relative costs and benefits of one of several management options. Unfortunately, often it is not possible to produce a precise quantitative estimate for a watershed ecological assessment because risks typically involve multiple stressors, from multiple sources, across a range of habitats. Instead, probability estimates can be expressed in high, medium, or low terms. For example, there is a high probability that shorebird populations in the immediate vicinity of a filled wetland will decline as a result of habitat elimination. Every effort should be made to state the likelihood of the outcome. Risk descriptions also can be expressed in relative terms, such as option A poses more risk to the assessment endpoint than option B. This is the kind of risk description derived from the RRM that ranks risks by habitat areas and stressors (Wiegers et al., 1998).

Because exposure to stressors is typically complex and possibly synergistic in watersheds, these relative data may be more useful in characterizing risks than single dose-response estimates derived from laboratory data. The nature and magnitude of predicted effects should be considered with regard to the structure and function of the affected ecological system, its resistance and resilience, spatial and temporal effects, indirect effects or cascading interactions, and the life-cycle of organisms.

The risk description interprets ecological significance, translating possible risk estimates into a discussion of their consequences for the watershed. The description may address the nature and magnitude of effects, the spatial and temporal patterns of effects, and the potential for ecosystem recovery if effects have already occurred. For instance, the factors limiting mountain whitefish survival and reproduction include elevated water temperatures, loss of lotic habitat, fluctuating water flows, and excessive sedimentation (U.S. EPA, 2002d). The evidence for these effects is demonstrated by evaluating favorable spawning conditions at two conditions in the river, comparison with habitat suitability indices, and review of the literature. The risks are described in Table 3.

The significance of the consequences of predicted effects may vary considerably for different types of ecological systems or habitat types. For example, in the Waquoit Bay assessment (U.S. EPA, 2002c), atmospheric deposition of nitrogen on loam soils far from a water body was deemed less of an impact than deposition occurring on sandy soils near an estuary. The effect of a herbicide may be quite different in a stream that derives most of its organic carbon energy from plants as compared with the effect in a stream that utilizes predominantly detrital-based organic carbon. The loss of wetland area may be highly significant if it represents the only habitat available in an area for waterfowl but may be negligible if it occurs among a much greater wetland area.

Sometimes risks can be described in relationship to a management option that is being considered. A review of historical data was used to describe the relationship between increased nitrogen loads and decreasing levels of eelgrass cover over time (U.S. EPA, 2002c). Managers then could select a particular percentage of eelgrass bed coverage as a management target. For example, if the restoration target was to support the growth of eelgrass on 30% of the main estuary, this would be comparable with 1970 conditions in Waquoit Bay. Thus, the nitrogen load would need to be reduced to 18,000 kg N per year (Figure 6).

In watershed assessments, the spatial distribution of exposure and effect is important to consider and can be assessed using GIS overlays of the two types of information. GIS maps can be helpful as analytical and communication tools. When large areas are parsed into small subunits or representative subsets are selected for analyses, key differences can be lost. GIS

Table 3. Middle Snake River case study: integration of stressors, responses, and recovery potential for the reproduction, growth, and survival of the mountain whitefish population

Factor	Stressor	Line of evidence	Risk	Uncertainty	Assumption	Recovery potential
Number of spawning fish	Loss of adult habitat, restrictions on movement to feeding and overwinter areas	LIT, BPJ	An increase in the population size is not possible with low or no reproduction	Low, very few adult fish are present in the Middle Snake River or nearby tributaries	Available fish surveys are a true indication of the scarcity of fish in this area	Low, without improving lotic conditions and lowering water temperature
Spawning	Water temperature too high for successful egg development, loss of lotic habitat	HSI, LIT, WQS, BPJ	Population cannot recover without successful reproduction	Low, water temperature requirements for spawning are known	Low spawning success attributed to poor water quality conditions	Low, the present annual water temperature regime does not support successful spawning
Fry	High water temperature, fluctuating water levels, loss of lotic habitat, and predation	HSI, LIT, WQS, BPJ	Population cannot recover without successful recruitment	Low, temperature requirements are known, but rearing habitat not identified	Fluctuating water levels push fry into deeper water where increased predation occurs	Low, the present annual water temperature regime does not support successful fry development
Food supply	Sedimentation and increased water temperature	LIT, Field, BPJ	Invertebrate fauna not adequate to support an increased mountain whitefish population	Moderate, adequate analysis of sampling information has not been completed	Food supply is limited	Low, without improving lotic conditions and lowering water temperature
Movement	Dams prohibit seasonal migratory movement by adults	LIT, Field, BPJ	Adults unable to reach upstream tributaries or refugia used for spawning or rearing	Moderate, limited information on migratory population in the Middle Snake River	Adult fish reported at Lower Salmon Falls Dam in the 1950s represented a migratory population	Low, without providing adequate fish passage facilities at the dams

BPJ = best professional judgment
Field = field survey in the Middle Snake River
HSI = habitat suitability index
LIT = literature search
WQS = water quality standards

Source: U.S. EPA (2002d).

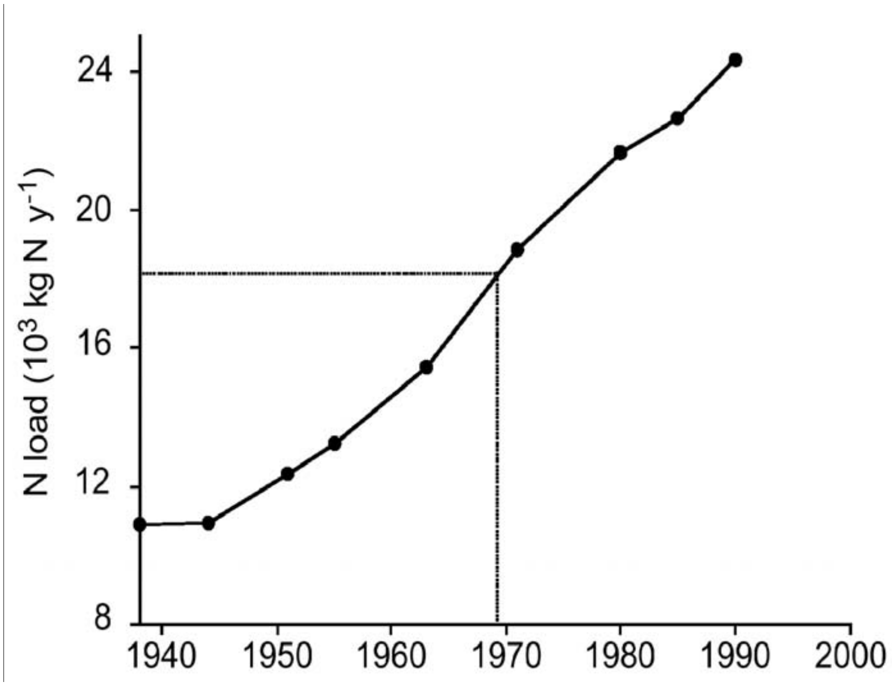

Figure 6. Historical changes in nitrogen (N) loading predicted by the nitrogen loading model. If the management target is a return to 30% eelgrass cover, then managers must reduce N loads to 18,000 kg N per year, loads comparable to those of around 1970.

Source: U.S. EPA (2002c).

approaches help address that problem by providing a mapping tool that can capture basic descriptive information and link calculations for subareas to produce an overall picture.

With a GIS, data layers can be added to identify other more specific potential sources of stressors, such as wastewater discharges and mining activities. Although certain types of land use may not be amenable to calculation on a percent area basis, they may be analyzed on a distance or density basis (i.e., how close a given mine is to aquatic life habitat or how many mines per square mile are within a certain area) (see Figure 7). Either of these calculations could provide useful estimates of mine "exposure." Before using GIS, ground-truthing of data is necessary.

The RRM projects described previously rely on GIS-based approaches to evaluate potentially important stressors in watersheds or regions. These approaches use maps to display major stressors and associated degree of potential biological impacts on maps. The maps greatly facilitate communication of the information to the public and managers. Stressors can also

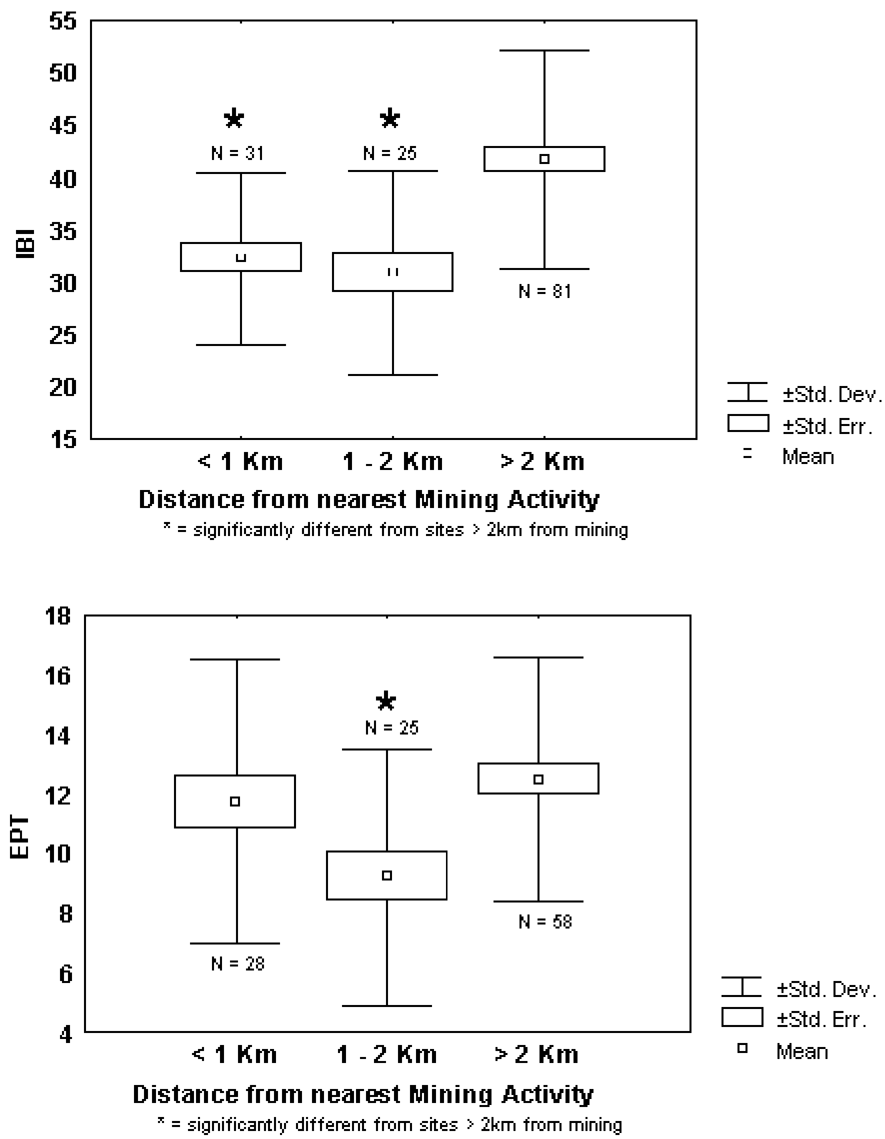

Figure 7. Fish index of biotic integrity (IBI) or insect ephemeroptera, plecoptera, and trichoptera (EPT) values in relationship to proximity to coal mining sources in the Clinch Valley ecological risk assessment. If the location of a source of potential stressors can be clearly defined (as in the case of point sources or those that can be spatially defined), it may be useful to relate measures of effect to the distance from the source (in this case active coal mines) to assess potential effects of a given type of source.

Source: U.S. EPA (2002a).

combine to influence the quantity and quality of habitat for species, and therefore habitat can be used as a basis for analyzing and showing how effects are integrated. Because habitat requirement vary by species, the range occupied by particular species, or biological communities needs to be considered. For example, land use impacts can be analyzed and shown according to the various habitats occurring across the landscape.

Besides GIS, graphs are also a helpful analytical and communication tool. In assessments where timing of events is critical (e.g., as in acid precipitation, or in the assessment of episodic events), graphs that show the timing and distribution of stressor events in comparison to an effects threshold may be a useful way to present the information. For prospective risk analyses supporting watershed planning, risk estimation derives the estimates of management measures for comparison between different management options. In EPA's TMDL protocols, this is referred to as a "linkage analysis" (U.S. EPA, 1999b). When exposure and effects data are limited or are not easily expressed in quantitative terms, qualitative evaluation techniques may be used to rank risks using subjective judgment and categories such as low, medium, and high.

It can be useful to incorporate results of risk analyses, and the probable co-occurrence of stressors and assessment endpoints, into the initial conceptual model. Pathways can be shaded or color-coded (e.g., green = no or few risks likely; yellow = risk possible; red = risk likely) on the conceptual model to help characterize the risk to assessment endpoints (Serveiss and Ohlson, 2006). This "updated" conceptual model could also be used to highlight where major uncertainties lie and/or additional sampling or analyses would be especially worthwhile.

Agreement among multiple lines of evidence increases the confidence in the conclusions, although any discrepancies warrant discussion. The Clinch Valley assessment used a table to show the various lines of evidence and to summarize the conclusions of risk and recovery potential to various life stages of mussels and fish (U.S. EPA, 2002a). In the Middle Snake River ecological risk assessment (U.S. EPA, 2002d), a water quality simulation model was used to predict attainment of water quality standards and habitat requirements for the life-cycle stages of coldwater fish (Table 3). Simulations were used to modify the model or provide information on uncertainty. The data from the model were supplemented with field data and literature review.

6.2. DESCRIBING UNCERTAINTY

There are two general sources of error in ecological assessments—uncertainty and variability. Uncertainty from lack of knowledge can be expressed in many ways during the analysis phase, such as insufficient data or spatial coverage of required data, inaccurate model inputs, GIS spatial resolution errors, and investigator bias (intended or unintended). Natural variability includes spatial and temporal heterogeneity in measures of interest, background ecological system factors (e.g., geology, precipitation), and natural disturbances (e.g., floods, fires, hurricanes), all of which can affect the ecological effects and relationships observed.

Uncertainties encountered throughout the assessment, including the basic data set, should be summarized in an uncertainty analysis (Suter, 1998; U.S. EPA, 1998a). There are likely to be many sources of uncertainty, such as measurement error (inappropriate, imprecise, inconsistent, or sparse measurements), conditions of observation (such as extrapolating from laboratory tests to field predictions), and limitations of models (oversimplifying complex ecological processes, incorrect assumptions). Uncertainties exist in every aspect of a risk assessment, especially in estimates of exposure, estimates of effects, and the integration of exposure and effect. The relationship between the measure of effect and the actual assessment endpoint is always subject to uncertainty. Watershed and regional assessments may contain more uncertainty than smaller scale assessments because of difficulties in obtaining sufficient data for multiple stressors, sources, and measures of effect for the entire geographic area.

Uncertainty can be described in either narrative form, with statistical distributions, or with sensitivity analysis. Narrative descriptions are especially suited for qualitative types of uncertainties, such as unclear communications, descriptive errors, and professional judgment and any assumptions made throughout the assessment. Depending on how data gaps are addressed, their associated uncertainties may be quantifiable. Natural variability and sampling error may be described statistically. Some uncertainties, such as those associated with modeling, can be addressed in terms of sensitivity analyses, in which certain model parameters (or inputs) are adjusted and the outputs compared (Suter, 1998; Hession et al., 1996). Other uncertainties, such as those associated with sparse data, cannot be reduced without additional data collection but may lead to identification of those data that greatly reduce uncertainties for future risk assessments. Uncertainties may increase if the appropriate temporal scale is not evaluated. Episodic pollution events, for example, can have dramatic, persistent ecological effects that might not be recognized using a short temporal horizon. In the Clinch and Powell ERA, episodic chemical spills from transportation or industrial accidents resulted in long-term reductions in native mussels downstream (Diamond and Serveiss, 2001). Long-term data sets, if available, are desirable to address such issues (Hunsaker et al., 1990).

The Middle Snake River assessment (U.S. EPA, 2002d) used Table 3 not only to summarize risk descriptions but also to describe the uncertainties and assumptions associated with each effect. In successive model simulations, input parameters were varied to provide information on uncertainty in model output. The data from the model were supplemented with field data and literature review. Sources of uncertainty included limited knowledge of stressor-effect relationships, variability in field measurements, model error, and estimation error. Each source of uncertainty was further described in narrative form.

When data are sufficient to proceed with a quantitative assessment, as when focusing on single pathways of the conceptual model (e.g., Serveiss et al., 2004; Hession et al., 1996), then quantitative measures of uncertainty are possible for the exposure and effect measures. Uncertainties associated with interactions among stressors and sources that are not considered in models only can be addressed qualitatively. RRM addresses multiple stressors and effects by using qualitative rankings in quantitative models. Uncertainty of model output can be represented quantitatively in sensitivity analyses, although the input is qualitative (Walker et al., 2001; Wiegers et al., 1998). Describing risk and uncertainty are two of the major features of risk characterization. Since there will always be uncertainty, it is important to have assessments peer-reviewed by other scientists in the field. Peer-review comments and a description of how the report was revised in response to comments should be made available.

6.3. RISK CHARACTERIZATION COMPONENTS

Risk assessors prepare a report for the risk managers using concepts outlined in Chapter 7, Risk Communication. Risk managers use this information to make decisions (see Chapter 8). Although these steps are presented chronologically in this report, these activities may occur throughout a watershed assessment. A watershed manager may need to make a relatively quick decision to control the impact of a spill and may ask the assessor for interim information before the assessment is completed. Thus, some elements of risk characterization and risk communication described in this and subsequent chapters may apply to the manner in which interim findings are presented earlier in the assessment.

Risk characterization can take the form of an isolated product by summarizing the earlier phases or it can be one part of the entire report. Either way, somewhere in the risk assessment report, the following information should be included:

- A description of risk assessor/risk manager planning results
- A discussion of the major data sources and analytical procedures used
- A review of the stressor-response relationships
- Estimates and descriptions of risks to the assessment endpoints
- A summary evaluation of the relative risk of different management scenarios for use as a basis for risk management decision making
- The major sources of uncertainty and the approaches used to address them
- Documentation of science policy judgments or default assumptions used to bridge information gaps and the basis for these assumptions

- Lines of evidence and the rationale for reaching the conclusions that were reached
- A description of the peer review process and a list of peer reviewers

Effective risk characterization must accurately translate the best available information about a risk into a language nonscientists can understand (NRC, 1996). The risk characterization should clearly communicate to the risk manager the major risks at some level of biological organization (organism, population, community, ecosystem), the ecological significance of the findings, and the level of uncertainty. It is important to clearly describe the ecological resources at risk, their value, and the costs of protecting (or failing to protect) the ecological resources. The ultimate success of risk characterization depends on successful risk communication.

7. RISK COMMUNICATION

Risk communication is the translation of scientific information in a form that is useful to managers and the public. Risk communication should occur throughout the assessment process, but ultimately what counts is how the results of the assessment are used. Hence, this is one of the latter chapters in this report.

When risk characterization is complete, a report is prepared showing estimates of risk. Clear presentation of the strengths, limitations (including uncertainties), and conclusions of the risk assessment will greatly enhance the assessment's usefulness in decision making (Serveiss, 2002). The challenges for risk communication include establishing a process for risk communication, targeting the audience, and communicating effectively.

7.1. THE RISK COMMUNICATION PROCESS

Regular and recurring communication between scientists and managers is needed to refine management objectives and to ensure that the risk assessment is relevant to the decision problems. The management objectives, and the relationships between management objectives, stressors, and management options, are often poorly defined at the start of the process (Reckhow, 1994). When the assessment is completed, the manager and key stakeholders should have a full and complete understanding of the assessment.

Involving stakeholders throughout the process increases the likelihood that findings will be useful and helps prevent confrontation or litigation. Watershed management requires an interactive, participatory approach (Glicken, 2000; U.S. EPA, 1995a), in which risk assessors both impart information to and gain information from stakeholders. In the ERA paradigm, stakeholders work as partners with scientific risk assessors at every stage of the process—defining the problem to be solved, prioritizing the risks posed, and evaluating and ranking the remedies (NRC, 1996).

Many types of risk cannot be assessed in a completely objective way because the evaluation of risk depends on the interpretation of goals and objectives, which is a value-laden process. A stakeholder-based decision process provides a means to make value judgments that are supported by the scientific evidence. The Delaware River TMDL for polychlorinated biphenyls was established as such a structured decision process. A Web site was used to describe the time and place for business and public meetings and the topics to be discussed (Delaware River Basin, 2006).

While stakeholder involvement is crucial, there are also reasons to refrain from involving all participants. An adequate treatment of science is possible in a stakeholder-driven process—but only if sufficient high-quality staff, time, and resources are available (U.S. EPA, 2001a). Where resources are not sufficient, or when a decision must be made quickly to address an imminent threat, stakeholder processes may not be the solution. The SAB also warns against handing off processes to a stakeholder group as a means to address mandates for which an agency has insufficient resources.

If stakeholder groups are too large (or intractable in their opinions or desires), it may be difficult to reach closure. There are likely to be different stakeholder groups with widely varying levels of interest in the process, and varying ability to commit time to the process. One solution appropriate for many risk assessments is to form a stakeholder advisory group consisting of a smaller group of well-respected and representative stakeholders (about 10 members). The advisory group serves as a communication pathway between the risk assessment team and the full set of interested parties in the community, serving both to identify values and concerns of the community at large and to communicate progress in risk assessment and risk management efforts back to the community.

An important challenge for the participatory approach lies in balancing meaningful stakeholder participation with the need to make progress in managing risks. Rather than allowing the stakeholders to have access to the risk management team and process at all times, it is important to strike a participation/management balance by providing well-defined, strategic time frames for stakeholder input.

7.2. ADDRESS THE TARGET AUDIENCE

Watershed assessments have multiple audiences and stakeholders. There are managers or groups with defined decision making authority, such as a governmental agency with regulatory authority over water quality or a county staffer charged with developing a land use plan. In addition, individuals or groups may actively participate in the watershed management process (e.g., watershed associations, river watch citizen groups, volunteer monitoring groups, industries, and agricultural associations). Other stakeholders include participants who are influenced by and have the ability to affect (if only through lifestyle and land use choices) the area of interest (e.g., land owners in the watershed). Communication needs differ accordingly with each stakeholder group. Glicken (2000) makes the further point that stakeholders or interested parties are identified and defined relative to specific issues. A characteristic of most watershed management exercises is that there are multiple issues to be addressed, and thus the participating stakeholders do not constitute a single, well-defined set but, rather, a complex network that shifts in relationship to issues. Efforts should be made to engage the participants on all issues and at every phase, as neglect may lead to alienation and contention (also refer to Section 3.1).

7.3. COMMUNICATE EFFECTIVELY

Risk characterizations should be prepared in a manner that is clear, transparent, reasonable, and consistent with other Risk Characterizations of similar scope prepared across programs in the Agency (U.S. EPA, 1998a). For effective risk communication, scientific information should be characterized in the manner most appropriate for addressing the major concerns. Since watershed risk assessments are place-based studies, maps are often important tools to aid in summarizing and conveying information (Serveiss, 2002). With the advent of powerful and accessible GIS and database tools, it has become feasible to analyze and display spatial information. Maps can provide easily grasped visual relationships between sources, stressors, and effects to which the public can relate. In the Clinch and Powell assessment, maps were especially useful in visualizing cumulative sources of stress in relationship to locations of threatened and endangered mussel species (U.S. EPA, 2002c). Such maps can then be used to evaluate vulnerability and prioritize management strategies.

Communication can be misleading if uncertainty is not described. At each step of the assessment process, what is not known should be presented along with what is known.

Clear presentation of the strengths, limitations, and conclusions of the risk assessment will greatly enhance the assessment's usefulness in decision making. Describing the association between a range of stressor levels and various risk estimates can be especially useful for managers who may need to choose among options with varying capabilities for stressor reduction. Successful risk communication leads to a higher likelihood of selecting optimal risk management actions.

8. RISK MANAGEMENT

The purpose of a watershed ERA is to provide managers with information to help address environmental risks (i.e., make risk management decisions). The goal of risk management is to take scientifically sound and cost-effective actions to reduce or prevent risks, while taking into account social, cultural, ethical, political, and legal considerations (PCCRARM, 1997).

8.1. EVALUATING MANAGEMENT ALTERNATIVES

Decisionmakers must select a particular management action, including no action. When environmental management goals and risk assessment endpoints are clearly defined and agreed upon by assessment participants, the risk assessment process is more likely to produce useful information for choosing an action.

Watershed management typically includes multiple objectives, and watershed degradation typically involves multiple sources and multiple risks. In an ideal case, the final management plan would be endorsed by all participants. However, when conflict arises between stakeholder groups with opposing goals, selecting appropriate management alternatives may be difficult. Some watershed projects may prove too controversial and reach a stalemate from time to time. In the worst cases, some stakeholders may choose to selectively ignore parts of the science or to withhold relevant information. Despite these challenges, some stakeholder involvement is always necessary (as discussed in Sections 7.1 and 7.2), especially before money is committed to possible solutions.

An important characteristic of many watershed risk management processes is the role of stakeholders in identifying and evaluating potential management options. It is generally not appropriate to use open stakeholder processes to make regulatory decisions that are explicitly mandated to agencies (U.S. EPA, 2001a). However, when selecting among management alternatives that are consistent with the law, stakeholder involvement increases acceptability of the outcome—particularly where social value judgments are required to choose the best level and approach for environmental protection.

Evaluation of the relative performance of candidate management options can be made on the basis of their abilities to achieve management objectives, usually through the use of an appropriate indicator or measurement endpoint. In some cases, this is a relatively straightforward process. For instance, the objective might be to reduce fish mortality from ammonia inputs, with an associated indicator of predicted un-ionized ammonia concentrations. If the modeling tools are present to evaluate un-ionized ammonia concentrations associated with each management alternative, then a direct comparison may be made. Designing assessment endpoints by relating them to management objectives enables them to be useful in risk management decisions.

Quantitative management objectives are always desirable and sometimes management options can be tested in a prospective risk analysis. In these instances, there may be a wide array of management options that can be considered (e.g., limit types of development, limit development density, use stormwater controls). In circumstances where there are no competing objectives, and if all possible management objectives can be ranked quantitatively using an agreed-upon scale, selecting the best management options is relatively straightforward. However, this rarely happens.

In most watershed management exercises, there are competing objectives and a lack of consensus. Objectives may be pre-established and unfocused (e.g., control future nutrient-related eutrophication in a lake). The management options also may change after the results of early analyses are presented.

When a quantifiable management objective may not be available or agreed upon by various stakeholder groups, a subjective ranking approach may be used. A simple value scoring approach that requires each stakeholder group to identify the best and worst of the candidate scenarios (combinations of management alternatives) can be used (Stewart and Scott, 1995). Best and worst scenarios are defined as the top and bottom of a subjective scale running from 100% to 0%, with intermediate scores relating to the relative "gaps" between the two well-defined extremes. The resulting scores obtained by a stakeholder group for a set of scenarios may then be displayed on a "thermometer" scale (Figure 8). In the figure, the stakeholder group ranked five scenarios, judging Scenario II the best and Scenario III the worst. A compilation of rankings submitted by multiple stakeholder groups would reveal the alternatives that are least objectionable to the participants.

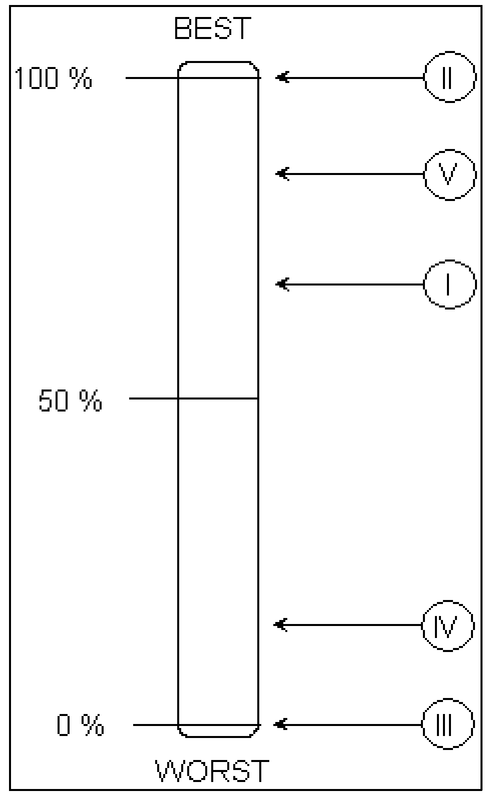

Figure 8. Thermometer scale for relative ranking of management options.

While the actual decision process is likely to vary on a case-by-case basis, some guidelines may help promote a successful outcome. First, as stated earlier, it is important to ensure that stakeholder groups are present at the table from the start of the process, particularly those with the potential to present obstacles to management. Second, the decision context and the role of stakeholders in the process should be made clear from the onset. Third, it is important for all parties to realize that evaluating alternatives is an iterative process and that scientific information that emerges during the process may require rethinking the nature and significance of risk pathways and, potentially, the management goals for a watershed. Finally, one must recognize that risk management may take time and money.

8.2. APPLICATION TO DECISION MAKING IN THE EPA CASE STUDIES

The three completed EPA case study reports each include a section on how the assessment assists with decision making. For the Clinch and Powell Valley assessment (U.S. EPA, 2002a), the ERA process advanced a better understanding of environmental problems. In particular, the conceptual model and multivariate analysis helped clarify the interrelationships between various components of the ecosystem and the manner in which human activities contribute to environmental problems within the watershed. In assessing environmental risk, a number of federal, state, and local environmental agencies and organizations came together to share data, explore and develop solutions, and undertake actions within the watershed. Risk assessment findings can also help direct the efforts of the Upper Tennessee River Basin Roundtable, which is composed of various individuals, agencies, and organizations that have an interest in protecting the watershed. ERA findings will be useful to the roundtable as it begins comprehensive strategic planning for watershed protection. Additionally, the numerous watershed coalition groups within the basin can use the risk assessment findings to direct their efforts to protect and improve water quality within their watershed.

The process of risk assessment helped lend further credence to what many professional resource managers had long conjectured about problems within the watershed, thereby providing more scientific support for taking actions to address identified problems. For example, there is now a better understanding of what sediment from cattle grazing contributes to the river. Such risk assessment findings will be useful to FWS and TVA personnel, who can now share this information with farmers and encourage them to take actions, such as building fences to keep cattle out of streams.

Examples of management actions that FWS and TVA have considered on the basis of the overall risk assessment findings include the following:

- Restoring additional abandoned mine lands throughout the watershed
- Further study of the chemical makeup of discharges from coal mining and processing facilities and the toxicity of these discharges to aquatic species
- Increasing the extent of forested riparian areas adjacent to and upstream of critical aquatic habitat sites for mussels and fish

- Implementing better spill control mechanisms on roadways and railroads near sensitive streams and more spill contingency plans for the watershed, which will enable the Virginia Department of Transportation and other agencies involved in constructing highway projects on or near waterways to design those projects to reduce catastrophic events and minimize impacts of accidental spills
- Installing BMPs for pasture and agricultural land to reduce sediment loading and implementing better treatment of wastewater discharges

In the Waquoit Bay ERA (U.S. EPA, 2002c), nitrogen loading from the watershed led to water quality problems and diminished populations of valued biotic resources. Nitrogen loading and estuarine loading models were developed as part of the assessment. These models helped managers to better understand nitrogen impacts and to make more informed decisions in addressing the problem. The ERA process also helped engage local citizens in the dialogue on mitigation strategies (including validating model predictions, long-term monitoring, and adaptive management).

Another valuable product of the Waquoit Bay ERA is the conceptual model, which allows citizens and local managers to view the watershed stressors and their associated ecological effects from a more holistic perspective. It emphasizes both point and nonpoint sources of pollution and shows how valued ecological resources are impacted by multiple stressors. Before formulation of the model, the focus was on analyzing single stressors or single species or on managing fisheries on a stock-by-stock basis. The conceptual model helped develop evaluation and management strategies that considered the wider effects of pollution on habitats.

EPA used simulation techniques developed for the middle segment of the Snake River (U.S. EPA, 2002d) and the associated conceptual model to determine impacts from a proposed new hydroelectric facility. The Federal Energy Regulatory Commission was convinced by EPA's risk assessment that cumulative impacts should be addressed when reviewing new or continued licensing for hydroelectric facilities. The assessment also provided community groups with a more robust analysis of the possible impact of impoundments on the river. Risk assessment participants were able to gain an improved understanding of the interrelationships between various components of the ecosystem and how human activities contribute to environmental problems within the watershed. For example, there is now a better understanding of the contribution of sediment to changes in the ecosystem, including eutrophication and decline of fish species. This information is useful to managers as they begin to regulate the sediment sources.

The simulation models developed for the Snake River can be used to test management scenarios. Those interested in land use activities and resource protection can better explore alternative management options for the watershed. The methods developed through this risk analysis can also be applied to other watersheds with similar conditions.

The benefits of watershed ERA for the three EPA-sponsored watershed assessments have been summarized and demonstrated above. Watershed ERA can also improve the organization and quality of several other types of watershed analyses, including TMDL analyses and use attainability analyses (UAAs), as described in the following sections.

8.3. APPLICATION TO TOTAL MAXIMUM DAILY LOAD PROCESS

The TMDL process includes activities similar to ERA risk analysis. For instance, the objective of source assessment is to “characterize the types, magnitudes, and locations of sources of sediment loading to the waterbody,” while the objective of linkage analysis is to “Define a linkage between the selected water quality targets and the identified sources to determine total assimilative capacity for sediment loading or total load reduction needed” (U.S. EPA, 1999b). The source assessment objective is consistent with exposure characterization, while the linkage analysis is similar to effects characterization.

An example of how watershed ERA can be applied to a TMDL watershed management exercise is the clean sediment TMDL for Redwood Creek, CA (U.S. EPA, 1999b), which was summarized in Serveiss et al. (2005). Redwood Creek had a designated use to support a coldwater fishery. The water body was listed as impaired on the state’s 303(d) list because salmonid stocks were declining. Upland sediment loading from forestry was believed to have salmonid impaired habitat quality. Changes to pool, substrate, and channel structure were apparent and also contributing to increased temperatures. Collectively, these changes affected the spawning and food production of salmonids. The TMDL listing and associated problem statement mimic ERA problem formulation activities.

Numeric water quality standards for sediment do not exist in California’s regulations, and the narrative criteria (support of the coldwater fishery) and a pollutant (clean sediment) made it necessary to select measures and target values for the analysis. The TMDL identified eight instream and seven “hillslope” indicators and targets. The instream indicators were measures of exposure, and they have been selected in a way that provides reasonable surrogates for effects. For instance, the first instream indicator was “Percent fines <0.85 mm in riffle crests of fish-bearing streams.” This indicator represents a critical node along a stressor-response pathway: that excess fine sediment clogs spawning gravels and results in reduced spawning success of salmonids. For the purpose of the analysis, the first instream indicator (and the seven other indicators) served as a surrogate for what would be the assessment endpoint (spawning success of salmonids) and thus was also used as a measure of effect. The hillslope indicators (e.g., number of road crossings with diversion potential) served as measures of exposure. Management (implementation) scenarios could then be compared and defined in terms of the hillslope indicators and their estimated effect on sediment loading.

Target values for the instream indicators were based on a combination of literature values, professional judgment, and comparison of impaired reaches within the watershed to unimpaired reference reaches, both within the watershed and in similar terrain. For instance, the target value for percent fines in riffle crests was determined to be less than 14%.

The linkage analysis for this TMDL was performed by estimating upland sediment loading rates for the impaired water body and the unimpaired reference streams. Based on this comparison, it was estimated that a 60% reduction in average annual sediment loading in Redwood Creek was needed to support uses. Specific allocations were made by comparing management scenarios and their impacts on erosion processes, which were mostly associated with specific land use activities and were expressed as long-term average annual loads per square mile. The individual load allocations were based on assessment of the controllability of different source categories, that is, the extent to which sources are associated with human activity and will respond to mitigation. The analysis indicated that the application of reasonable reductions to the controllable load would be adequate to meet the TMDL.

The Redwood Creek TMDL did not follow all the ERA steps because of time and budget constraints. Yet, this TMDL has most of the components of a watershed risk assessment, including problem formulation; identification of environmental management goals, options, and objectives; assessment endpoints and associated measures; and the equivalent of exposure characterization and effects characterization. In other words, the TMDL was performed in a manner consistent with the watershed ERA approach. The scientific underpinnings of TMDL development can be improved using several features of watershed assessment (Serveiss et al., 2005).

8.4. APPLICATION TO THE UAA PROCESS

Beneficial uses of water bodies are designated by states under the Clean Water Act. Common beneficial uses include warm or coldwater aquatic life uses, which are ecologically based goals of a watershed that are subject to enforcement, if necessary. The designated use dictates the physicochemical water quality standards that need to be maintained in a water body. Therefore, proper identification of uses is a critical component of all water quality programs. A UAA is defined as the process of determining attainable beneficial uses, or modifying designated uses that are inappropriate (U.S. EPA, 1994). The Federal Water Quality Standards Regulation, 40 CFR 131.10(g), defines a UAA as "a structured scientific assessment of the factors affecting the attainment of a use which may include physical, chemical, biological, and economic factors."

A UAA consists of three general steps: (1) identifying and defining existing proposed or designated beneficial uses that were present as of November 1975, (40 CFR 131.3(e)); (2) determining whether uses are appropriate for the water body, and (3) identifying constraints on these uses by examining physical, chemical, and biological characteristics of the water body as well as socioeconomic constraints. The Federal Water Quality Standards Regulation defines six factors, any one of which, if demonstrated, could be used to remove an inappropriate use or establish subcategories of a use (see below). The documentation necessary to support any of these factors has been unclear and a source of confusion in UAAs to date, limiting the use of this essential tool. This is especially true for substantially modified watersheds (e.g., urban or agricultural watersheds and streams that are effluent dominated) in which "human caused conditions" predominate (see item 3 in list below). The following six factors are associated with modifying or removing a 101(a) use within a use attainability analysis (UAA), as cited in 40 CFR 131.10(g):

1. Naturally occurring pollutant concentrations prevent attainment of the use.

2. Natural, ephemeral, intermittent, or low-flow conditions or water levels prevent the attainment of the use.

3. Human-caused conditions or sources of pollution prevent attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place.

4. Dams, diversions, or other types of hydrologic modifications preclude attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in attainment of the use.

5. Physical conditions related to the natural features of the water body, such as the lack of a proper substrate, cover, flow depth…unrelated to water quality preclude attainment of aquatic life protection uses.

6. Controls more stringent than those required by Sections 301(b)(1)(A) and (B) and 306 of the Clean Water Act would result in substantial and widespread economic and social impact.

The watershed ERA framework offers many advantages for conducting UAAs, because: (1) stakeholders in the watershed (including resource agencies and wastewater dischargers) participate in defining the assessment process; (2) the assessment explicitly treats multiple stressors, including physical and biological, and other water quality stressors; and (3) the assessment is necessarily place based because it deals with a specific watershed or basin. UAAs have been thwarted in many cases because it was often difficult to reach consensus on the types of information needed and how multiple lines of evidence should be interpreted. Watershed ERA can greatly improve the performance and transparency of UAAs by providing the scientific structure necessary to evaluate relevant information in a comprehensive and efficient manner.

Figure 9 shows a generalized watershed ERA process that lends itself well to UAAs. Using the watershed ERA approach, the UAA methodology relies on the identification and evaluation of specific indicators (i.e., assessment endpoints) and measures that represent salient characteristics of a given use (i.e., measures of effect) as defined by the standards. Through objective analysis of these measures (i.e., risk analysis), and comparison of those measures with known minimum criteria, thresholds, or requirements needed to support the use, an assessment is made as to the attainability of that use (i.e., risk characterization). Explicit in these analyses is a determination of specific water body attributes that are preventing attainability of a given use (stressor identification evaluation or causal analyses). These attributes are evaluated to determine whether certain modifications or controls would allow the use to be attainable and, if so, the feasibility or reasonableness of those options (i.e., risk management).

The first step in a UAA is problem formulation, which includes determination of the assessment endpoints indicative of the use in question, and conceptual model development. Assessment endpoint selection (as well as measures of effect) should be identified using a stakeholder process as described earlier in this document. Because UAAs depend on at least one of the six factors to be legally defensible, these factors become central to conceptual model development. In practice, socioeconomic constraints (item 6 listed above) are handled differently from ecological constraints (items 1 through 5 listed above) and are addressed outside the ERA process. The remaining five natural or anthropogenic factors then become categories of stressors that help define relevant potential stressors and their sources in the watershed. The assessment endpoints identified during problem formulation give rise to specific measures of effect that can be analyzed and compared with what is minimally needed to maintain that endpoint in the water body. An example endpoint would be salmonid abundance and distribution for a beneficial use that identifies coldwater aquatic life habitat.

Step 1: Problem Formulation:

- Translate proposed use into a goal and objectives.
- Develop conceptual model.
- Identify assessment endpoints and measures of effect.

Step 2: Risk Analysis

- Evaluate stressors, pathways, and measures of effect.
- Use reference conditions and watershed data to determine effects on assessment endpoints.

Step 3: Risk Characterization

- Compare stressor levels and physicochemical regimes with minimum thresholds or criteria for assessment endpoints.
- Determine relevant causes of nonattainability; stressor identification evaluation.

Step 4: Risk Management

- Identify control options if applicable.
- Consider feasibility of meeting use given candidate causes.
- Conduct stakeholder discussions and management option evaluation.
- Modify use if appropriate.

Figure 9. General steps involved in conducting a use attainability analysis (UAA) using a watershed ecological risk framework.

In a UAA context, risk is analyzed in terms of the ability of an indicator or assemblage maintaining itself given the five ecological factors as they exist in the water body. Later steps in risk characterization and risk management will determine whether present constraints in the water body can and should be remediated to better meet the use.

A UAA of two small central valley streams in California followed a watershed ERA framework. The conditions suitable to support coldwater ecosystems (COLD) include preservation or enhancement of aquatic habitats, vegetation, fish, invertebrates, or wildlife. Both salmonid habitat suitability indices and available habitat information for coldwater stoneflies were used to assess suitability of the COLD use. Measures were compared with criteria or "threshold values" to determine whether each indicator is supported and the use is attainable.

Using published habitat suitability parameters for salmonids, a simple conceptual model was constructed to delineate potentially important pathways for determining attainability of the COLD use with respect to fish. Physical habitat and chemical data indicative of habitat requirements for assessment endpoints were collected and combined with available literature information for the region to identify likely stressors affecting the maintenance of obligate cool- or coldwater aquatic life. The relative strength of each cause-effect relationship was then determined by considering the degree to which the indicator measurements differed from minimum thresholds needed to maintain coldwater fauna. For aquatic life uses, such as COLD, comparisons between measured values and thresholds could be quantitatively determined in part using fish HSIs and their various component values. The greater the disparity between the measured HSI component value for a given environmental factor (e.g., sediment particle size) and the minimum threshold, as defined by the HSI, the more limiting that particular factor is in preventing attainment of the use. These quantitative assessments were often supplemented with qualitative evaluations.

A decision tree was developed that presents the process used to determine whether the COLD use is attainable. If analyses indicated that water body conditions do not meet minimum thresholds for a particular use, the causes for nonattainment were then evaluated. The poorer a particular factor actually is, in comparison with minimum threshold criteria, the more limiting that factor is in attaining the use and, therefore, the less likely it is that the use is attainable. A weight-of-evidence approach was used to synthesize results of the conceptual model approach to determine overall attainability of the COLD use. In this example, indicator results (i.e., trout and stonefly results for COLD use) were consistent. More consistency in results among indicators (e.g., all indicators demonstrate a low probability of beneficial use attainment) suggests a higher degree of certainty in the overall assessment regarding use attainability. The more factors that do not meet minimum threshold criteria, the more certain that the use is not attainable under current conditions.

Using a watershed ERA framework, regulatory and resource agencies were able to quickly and efficiently weigh management options and the feasibility of making the COLD use attainable in these streams. These UAAs confirmed that COLD use was not attainable in the streams examined, given current hydrological and physical conditions.

9. CONCLUSIONS

This report advocates integrating the watershed approach with ecological risk assessment to make more informed environmental management decisions. This integration offers numerous benefits for those faced with the challenge of structuring environmental monitoring and analysis efforts and using such information for decision making. The process fosters regular interactions between scientists and managers to increase the likelihood that the optimal suite of scientific data is collected, analyzed, and considered in decision making.

Developing a management goal for a watershed ERA encourages organizations and the community to work together to develop a common vision, share information, and understand problems and ecological concerns. The usefulness of a risk assessment is enhanced when scientists and managers communicate regularly throughout the process. Agreeing on the focus, scope, and complexity of an assessment increases the likelihood that the collected and analyzed data will be of value to and used by managers.

The analysis phase of risk assessment addresses, to the degree technology and resources permit, the spatial and temporal co-occurrence of exposure of ecological resources to stressors and the direct and indirect effects of such exposures to valued ecological resources. Based on a few watershed ecological risk assessments conducted to date, risk analysis is the most challenging phase of the process because there are typically multiple sources and stressors that may vary over time and space. Tools such as GIS and CADDIS can assist with deciphering causes of ecological degradation. Often it is useful to simplify the analytical approach in a watershed setting by focusing on the impacts of one stressor or to use multivariate analysis to describe associations between sources or stressors and effects. Risk characterization integrates multiple lines of evidence into a clear summary of the risks to ecological resources. The risk characterization may predict the magnitude of changes in environmental quality associated with alternative management actions. It also identifies critical knowledge gaps and describes the level of certainty of the findings.

Interim findings need to be presented to managers to provide a clear view of the significance of the findings and to enable conceptualizing and possibly implementing potential solutions. The analysis can also be refined based on new information. Maps, graphs, and tables are among the best analytical tools for explaining relationships between stressors and effects. The presentation needs to be targeted for a specific audience, and more than one final presentation format may be necessary.

Documenting the impacts to valued ecological resources and the importance of their protection helps justify management actions. Even without direct regulatory authority, environmental improvements can be attained because the increased awareness that occurs as a result of documenting the consequences of human activities on the environment will encourage positive behavioral changes and management actions. In addition, improvements in the communication and coordination associated with the process can reduce duplication of effort.

Elements of the watershed ERA process can be used alone or in conjunction with other analytical decision making frameworks (e.g., TMDLs and UAAs). Scientists, managers, and stakeholders should communicate to determine the goals, objectives, focus, scope and complexity of an assessment. Developing a conceptual model requiring an interdisciplinary team of scientists, preferably with some local knowledge helps to better understand and communicate how potential stressors might affect valued ecological resources. The analysis plan helps to focus the assessment and to determine which data to collect and analyze. It should be based on perceived importance of stressors on assessment endpoints, corrective action potential, and resource constraints. The analysis plan should be considered an iterative process, where modifications are made as results are obtained.

Scientists, managers, and stakeholders should meet periodically to discuss interim findings and modify or refine the thrust of analysis efforts based on intermittent deliberations as necessary. With multiple stressors and multiple pathways, multivariate analyses or categorical ranking schemes may be needed to compare land use, stressors, and biotic measures. Conclusions need to be based on integrating multiple lines of evidence and discussing the degree of uncertainty in the findings.

Presentation of assessment findings needs to be targeted for particular audiences. As a result, more than one report may be needed. Maps, GIS, and other visual-based graphics are often very useful for communicating risk assessment results. All these steps will increase the likelihood that environmental monitoring and assessment data will be used in decision making.

GLOSSARY

The following definitions are derived from EPA's Watershed Academy training module on watershed ecological risk assessment (U.S. EPA, 2000b), *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998a), and Terminology Reference System (http://www.epa.gov/trs/index.htm).

Adverse ecological effects – Changes that are considered undesirable because they alter valued structural or functional characteristics of ecosystems or their components. An evaluation of adversity may consider the type, intensity, and scale of the effect as well as the potential for recovery.

Assessment endpoint – An explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes. For example, salmon are valued ecological entities; reproduction and age class structure are some of their important attributes. Together salmon reproduction and age class structure form an assessment endpoint.

Characterization of ecological effects – A portion of the analysis phase of ecological risk assessment that evaluates a stressor's ability to cause adverse effects under a particular set of circumstances.

Characterization of exposure – A portion of the analysis phase of ecological risk assessment that evaluates the interaction of the stressor with one or more ecological entities. Exposure can be expressed as co-occurrence or contact, depending on the stressor and ecological component involved.

Coal fines – Fine particulate manner from coal mining operations.

Comparative risk assessment – A process that generally uses an expert judgment approach to evaluate the relative magnitude of effects and set priorities among a wide range of environmental problems. Some applications of this process are similar to the problem formulation portion of an ecological risk assessment in that the outcome may help select topics for further evaluation and help focus limited resources on areas with the greatest risk reduction potential.

Conceptual model – A conceptual model in problem formulation for ecological risk assessment is a written description and visual representation of predicted relationships between assessment endpoints and the stressors to which they may be exposed. The conceptual model also describes the sources of stressors, the ecosystem potentially at risk, the relationships between measures of effect and assessment endpoints, and exposure scenarios.

Dose-response – How a biological organism's response to a toxic susbstance quantitatively shifts as its overall exposure to the substance changes (e.g., a small dose of carbon monoxide may cause drowsiness; a large dose may be fatal).

Ecological risk assessment – A process for analyzing environmental problems intended to increase the use of ecological science in decision making. The process evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors.

Exposure – Co-occurrence of or contact between a stressor and a receptor.

Exposure pathway – The course a chemical or physical agent takes from a source to an exposed organism. Each exposure pathway includes a source, an exposure point, and an exposure route. Air or water may be transport/exposure media if the source and endpoint are not in contact.

Lines of evidence – Information derived from different sources or by different techniques that can be combined to reduce uncertainty and increase defensibility of a finding.

Linkage analysis – Terminology used in EPA's total maximum daily load (TMDL) protocols to describe a quantitative analysis of stressor response, typically accomplished with a simulation model.

Management scenario – A structured combination of management options selected for evaluation.

Measure of effect – A change in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed.

Measure of exposure – A measure of stressor existence and movement in the environment and its contact or co-occurrence with the assessment endpoint.

Problem formulation – The first phase of ecological risk assessment, which includes a preliminary description of exposure and ecological effects, scientific data and data needs, key factors to be considered, and the scope and objectives of the assessment. This phase produces the risk hypotheses, conceptual model, and analysis plan, which serve as the basis for the assessment.

Receptor – The ecological entity that is exposed to the stressor.

Risk analysis phase – A phase of ecological risk assessment consisting of two main parts: (1) characterization of exposure—evaluating the interaction of the stressor with one or more ecological entities, and (2) characterization of ecological effects—evaluating the ability of a stressor(s) to cause adverse effects to an assessment endpoint under a particular set of circumstances.

Risk characterization phase – A phase of ecological risk assessment that integrates the exposure and stressor response profiles to evaluate the likelihood of adverse ecological effects associated with exposure to a stressor. Lines of evidence and the adversity of effects are discussed.

Risk estimation – Ideally, the conclusions of the risk characterization phase preferably expressed in a quantitative manner (e.g., there is a 20% chance of 50% mortality under the circumstances assessed), but also if expressed as a qualitative statement (e.g., there is a high likelihood of mortality occurring).

Risk hypothesis – A candidate description of the relationship between a particular stressor and an assessment endpoint. A risk hypothesis formulates proposed answers to questions risk assessors have about what responses assessment endpoints will show when they are exposed to stressors and how exposure will occur.

Risk management – The process of evaluating and selecting action alternatives in response to risk assessment findings. The goal of risk management is to recommend scientifically sound, cost-effective actions that reduce or prevent risks while taking into account social, cultural, ethical, political, and legal considerations.

Seascape – The marine equivalent of landscape ecology and tied into the designation of special areas as marine protected areas.

Source – An action that releases to the environment or imposes on the environment a chemical, physical, or biological stressor or stressors.

Stakeholder – An individual or group influenced by, and with an ability to significantly affect (either directly or indirectly), the topical area of interest.

Stressor – Any physical, chemical, or biological entity (agent) that can induce an adverse response.

Stressor-response curve – A graphic, quantitative representation of the relationship between a stressor (such as a pesticide concentration in the water column) and an ecological effect (such as mortality of a given fish species if exposed to a certain concentration of that pesticide).

Tiered approach – An approach to assessment that commences with simple scoping analyses and proceeds to additional tiers of more complex analyses as needed based on the results of earlier tiers or information that becomes available as the assessment progresses.

Watershed approach – A framework for coordinating environmental management that focuses public- and private-sector efforts on addressing the highest priority problems within hydrologically defined geographic areas, taking into consideration both ground-water and surface-water flow.

Watershed ecological risk assessment – An ecological risk assessment carried out at the watershed level, usually in support of watershed management efforts.

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